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Waterways Experiment Station

Wetlands Research Program Technical Report WRP-RE-6

Influence of Hydrologic Loading Rate on Phosphorus Retention and Ecosystem Productivity in Created Wetlands

by William J. Mitsch, Julie K. Cronk

January 1995 – Final Report
Approved For Public Release; Distribution Is Unlimited

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	Task		Task
CP	Critical Processes	RE	Restoration & Establishment
DE	Delineation & Evaluation	SM	Stewardship & Management

Wetlands Research Program

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by William J. Mitsch, Julie K. Cronk

School of Natural Resources
The Ohio State University
Columbus, OH 43210

Final report

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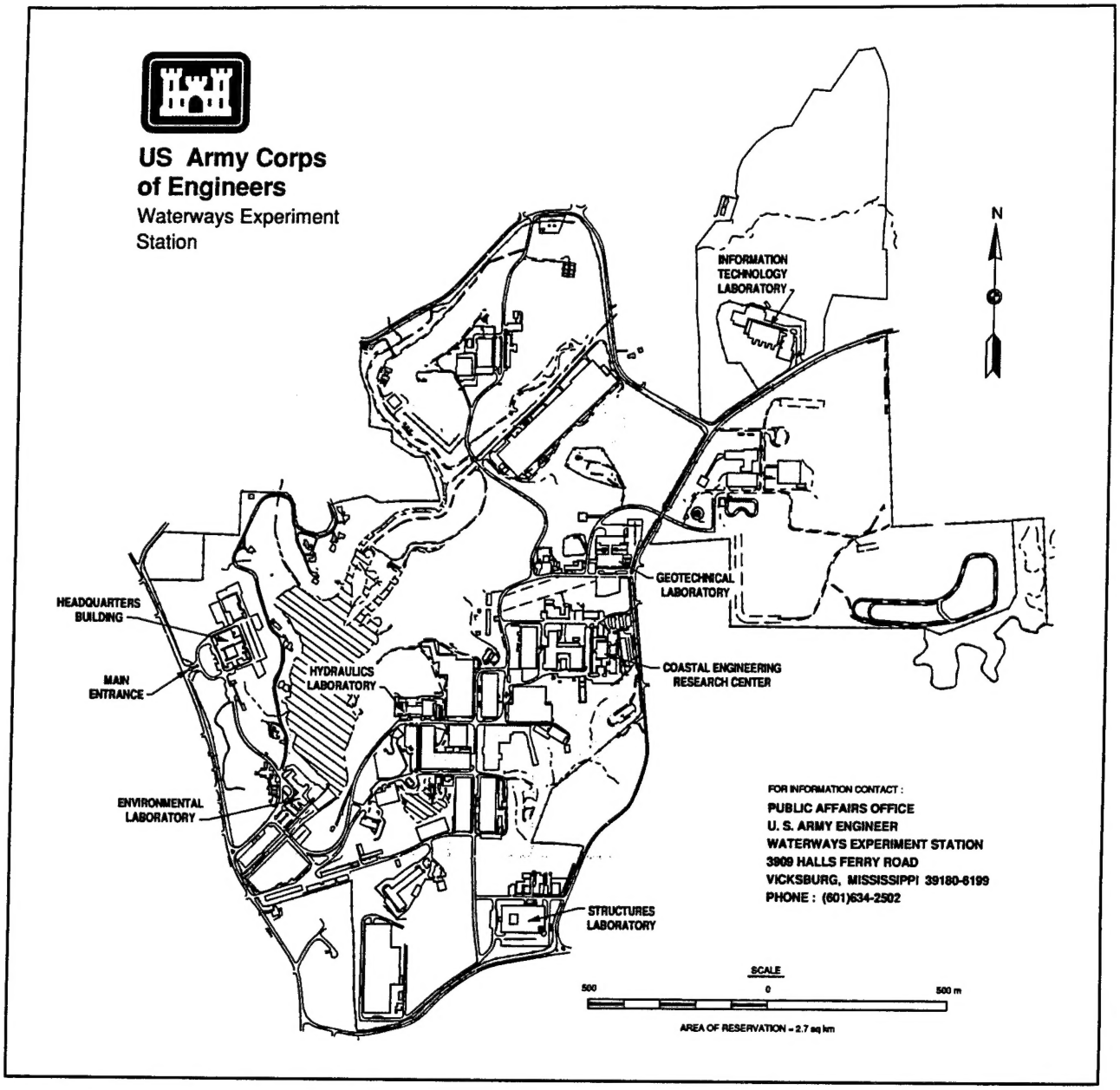

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Function of Created Wetlands

Influence of Hydrologic Loading Rate on Phosphorus Retention and Ecosystem Productivity in Created Wetlands (WRP-RE-6)

ISSUE:

An important aspect of designing constructed wetlands is to decide the optimum amount of water that can be discharged into the wetland. A hydrologic loading rate that is too high may overload the system, thus compromising the ability of the system to retain or transform nutrients. Additionally, other wetland functions, such as primary productivity and water column metabolism, may be related to hydrologic conditions.

RESEARCH:

Four 2- to 3-ha wetlands were constructed in Lake County, Illinois, and flooded with water from the adjacent Des Plaines River. Two of the wetlands were "low-flow wetlands" receiving 10 to 15 cm of water per week. Two others were "high-flow wetlands" receiving 34 to 38 cm of water per week. Measurements of phosphorus, macrophytes, periphyton, and water column productivity were made over a 2- to 5-year period.

SUMMARY:

All four wetlands retained significant amounts of the incoming phosphorus; phosphorus was retained closer to the inflow in the low-flow wetlands. Macrophyte productivity did not appear to be directly influenced by flow rate, while periphyton and water column production was generally highest in the high-flow wetlands.

AVAILABILITY OF REPORT:

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About the Authors:

Dr. William J. Mitsch, Wetland Ecologist, is a Professor at the School of Natural Resources at The Ohio State University. Dr. Julie K. Cronk, Wetland Ecologist, is Assistant Professor of Agricultural Engineering at University of Maryland, College Park. Point of contact at WES is Barbara A. Kleiss, phone (601) 634-3836.

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Preface

The work described in this report was authorized by Headquarters, U.S. Army Corps of Engineers (HQUSACE), as part of the Protection, Restoration, and Establishment Task Area of the Wetlands Research Program (WRP). The work was performed under Work Unit 32761, "Wetland Field Demonstrations" for which Dr. Mary C. Landin was the Technical Manager. Ms. Denise White (CECW-ON) was the WRP Technical Monitor for this work.

Mr. David Mathis (CERD-C) was the WRP Coordinator at the Directorate of Research and Development, HQUSACE; Dr. William L. Klesch (CECW-PO) served as the WRP Technical Monitors' Representative; Dr. Russell F. Theriot, U.S. Army Engineer Waterways Experiment Station (WES), was the Wetlands Program Manager; and Dr. Mary C. Landin was the Task Area Manager.

This work was performed by Drs. William J. Mitsch and Julie K. Cronk, under Contract No. DACW3991C0071. Dr. Mitsch is a Professor and Dr. Cronk was then a Post-Doctoral Researcher at the School of Natural Resources, The Ohio State University. The work was performed under the direct supervision of Ms. Barbara A. Kleiss, Research Ecologist, Wetlands Branch (WB), Ecological Research Division (ERD), and under the general supervision of Mr. Carl E. Brown, Chief, WB, Dr. Conrad J. Kirby, Chief, ERD, and Dr. John W. Keeley, Director, EL.

The authors would like to acknowledge contributions to the report from the following: Xinyuan (Ben) Wu, Post-Doctoral Researcher; graduate students Joseph Ely, M. Sioban Fennessy, Neal Flanagan, Robert W. Nairn, and Naiming Wang; and undergraduate assistants Jonathan Green and Douglas Stuart.

Additional funding, particularly from the U.S. Environmental Protection Agency, Chicago and Duluth, provided earlier data gathering and modeling support. Support and collaboration were received from Wetland Research, Inc., particularly Dr. Donald L. Hey and Dr. Dan Mason, and from members of the Technical Advisory Committee for the Des Plaines Project, especially Drs. Bob Kadlec, Arnold van der Valk, Bill Crumpton, and Hank Sather.

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1 Overview

Introduction and Site Conditions

The report herein represents the results of The Ohio State University's research program at the Des Plaines River Wetland Demonstration Project, Lake County, IL, for the period April 1991 through December 1992. The experimental wetlands at the Des Plaines River Wetland Demonstration Project were designed to develop and test wetland design principles, construction methods, and management programs needed to create and maintain wetlands for the purposes of water quality management, flood control, and fish and wildlife habitat (Hey et al. 1989). Specifically, this report summarizes research results and analysis for the following aspects of the Des Plaines River experimental wetlands:

- Phosphorus Dynamics and Retention.
- Macrophyte Productivity and Succession.
- Periphyton Productivity.
- Water Column Dissolved Oxygen and Metabolism.

The overall experimental conditions for the wetlands were established to investigate the role of hydrologic flow-through conditions on all of the above conditions in newly constructed freshwater marshes of the Midwest. Where appropriate, data from previous years are incorporated for comparison with the new data collected as part of this research. In other cases, additional analysis is provided for earlier data. Because water was first pumped on a sustaining basis to the Des Plaines River experimental wetlands in September 1989, the results presented here provide the status of flow-through constructed freshwater marshes after 3 years of existence.

Des Plaines River Wetland Site

The site is located 80 km north of Chicago in the town of Wadsworth, IL (Figure 1), and it includes 4.5 km of the upper Des Plaines River and 180 ha of riparian land. The Des Plaines River drains a watershed of approximately 545 km^2 (80 percent agricultural and 20 percent urban). High levels of suspended solids (60 to 100 $mg \cdot L^{-1}$) are a primary water quality concern, and total phosphorus levels are considered moderate to high for river quality (from 100 to 200 $\mu g\text{-}P \cdot L^{-1}$).

Experimental Wetlands 3 - 6 (Figure 2) were constructed on the Des Plaines River wetland site and were used in the research described herein. Bathymetric maps of the four wetlands (Figures 3-6) indicate sampling sites used for studies described in this report. The maps indicate location of permanent quadrats for diurnal oxygen and macrophyte vegetation measurements, periphyton samplers, inflows, and outflows discussed in this report. Depth measurements made in 1990 are also shown.

The experimental wetlands are hydrologically isolated from one another. A pump station was installed on the river to deliver known amounts of river water to each wetland. The system consists of an intake structure and pump house, two submersible pumps (propeller type), each of which is rated at 0.7 cu m/sec (25 cfs), and 2,200 m of low pressure concrete and ductile iron pipe to carry water to the wetland basins. Water inflow was monitored by acoustic velocity meters, and the outflow of the experimental wetlands was determined by weir equations.

Detailed hydrologic budget data collection by project staff began October 1, 1989. Water flow conditions for the four experimental wetlands are summarized in Table 1 and Figures 7 - 10. Wetlands 3 and 5 were high -flow wetlands (HFW), with flow generally averaging 34 to 38 cm/week (except for unusually highflow in Wetland 3 in 1992). Wetlands 4 and 6 were low-flow wetlands (LFW), with flow generally averaging less than 10 to 15 cm/week. Flow conditions in 1989-90 were erratic because of start-up difficulties. Flow conditions and hence water levels were more consistent during the 1991 growing season. In 1992, flow to the wetlands varied according to flow in the Des Plaines River. Details of water budgets and basin construction are given by Hey, Barrett, and Biegen (1994).

The pumping schedule was not consistent, with several periods when pumping was discontinued for at least 1 week. Pumping was done on a Monday through Friday schedule with pumps turned on in the morning and off in the late afternoon. No water was pumped into the wetlands on weekends, and except for the winter of 1989-90, pumping was stopped during the winters. Thus conditions described in this report are primarily for growing season conditions in constructed wetlands.

Other summaries and studies of research results of the Des Plaines River experimental wetlands are presented by Christensen (1991), Fennessy (1991), Hensel and Miller (1991), Kadlec and Hey (1991), Cronk (1992), Sather (1992a, b), Mitsch (1992), Cronk and Mitsch (in press), and Mitsch et al. (in press). A special issue of *Ecological Engineering* (Sanville and Mitsch 1994) features this and other concurrent research at the Des Plaines River site.

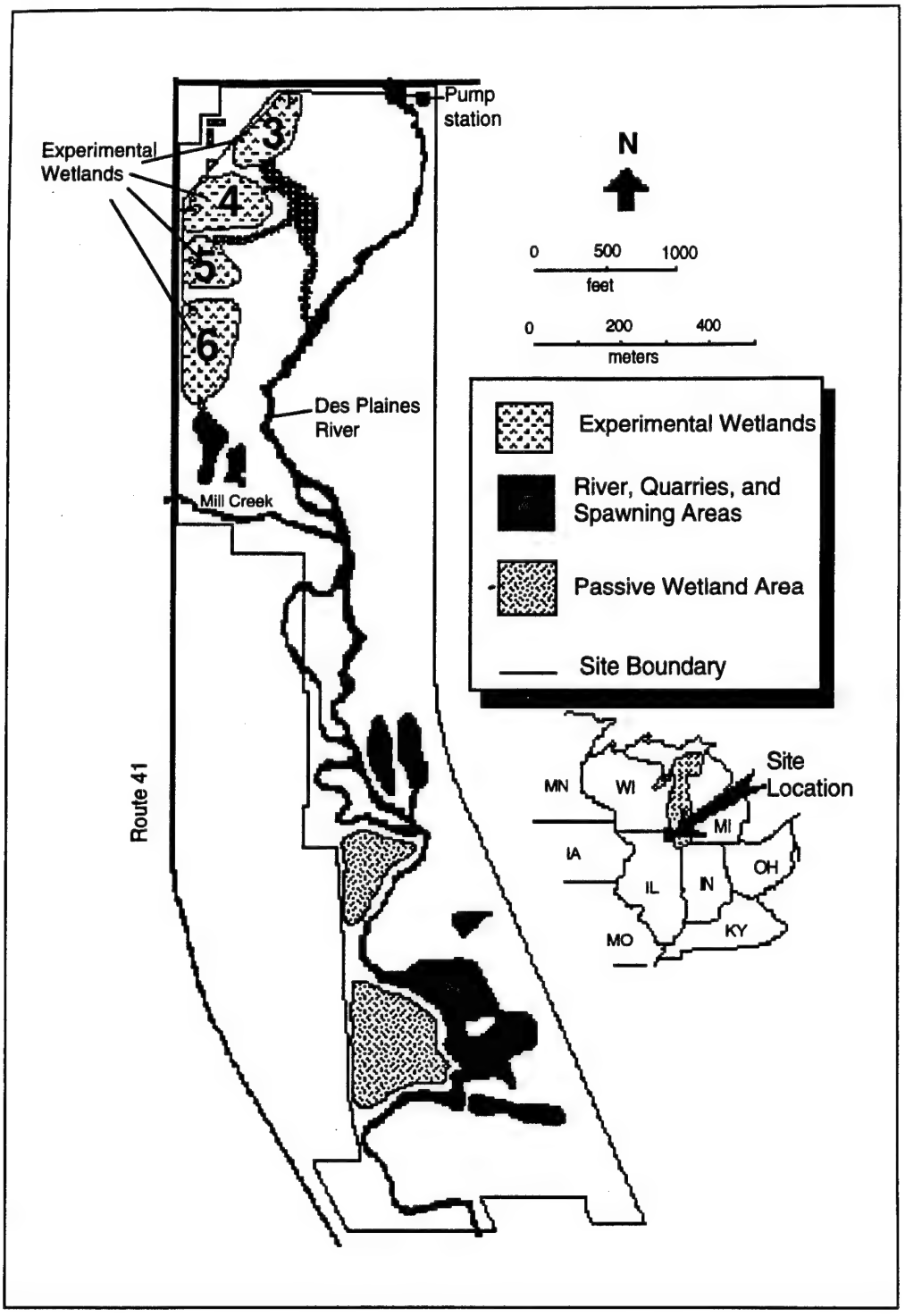

Figure 1. Des Plaines River Wetlands Demonstration Project near Wadsworth, IL

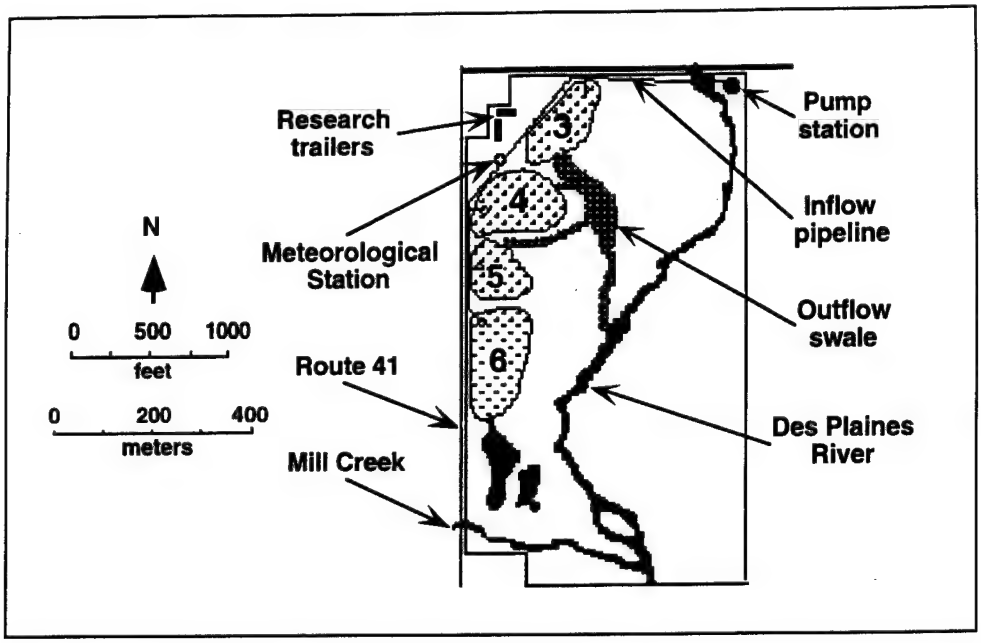

Figure 2. Layout of four experimental wetlands

Table 1
Hydrology of Des Plaines River Wetland Demonstration Project (October 1989 - September 1992)

	Wetland 3 (HFW 3)	Wetland 4 (LFW 4)	Wetland 5 (HFW 5)	Wetland 6 (LFW 6)
Size, acres	5.75	5.79	4.61	8.53
Size, ha	2.33	2.34	1.87	3.45
Design Inflow, cm/week	30	5	30	5
October 1989 - September 1990				
Inflow, cm/week (n = 52)	37.8 ± 2.2[a]	9.8 ± 0.9	35.2 ± 2.2	16.1 ± 1.8
Outflow, cm/week (n = 52)	38.5 ± 1.9	9.0 ± 1.3	33.9 ± 2.0	4.1 ± 0.9
Precipitation, cm/week	1.6	1.6	1.6	1.6
Evapotranspiration,cm/week	1.9	1.9	1.9	1.9
April 1991 - September 1991				
Inflow, cm/week (n = 26)	38.0 ± 3.7	7.4 ± 1.6	37.7 ± 3.7	9.1 ± 1.4
Outflow, cm/week (no. of weeks)	34.8 ± 3.3(26)	5.3 ± 0.8(22)	23.6 ± 2.6(19)	2.8 ± 0.8(5)
Precipitation, cm/week	1.5	1.5	1.5	1.5
Evapotranspiration, cm/wk	3.4	3.4	3.4	3.4
April 1992 - September 1992				
Inflow, cm/week (n = 25)	97.1 ± 23.5	13.7 ± 3.1	34.3 ± 10.0	---
Outflow, cm/week (n = 25)	92.1 ± 23.4	9.4 ± 3.2	25.3 ± 9.7	---
Precipitation, cm/week	1.6	1.6	1.6	1.6
Evapotranspiration, cm/week	3.4	3.4	3.4	3.4

[a] Numbers are average ± standard error.

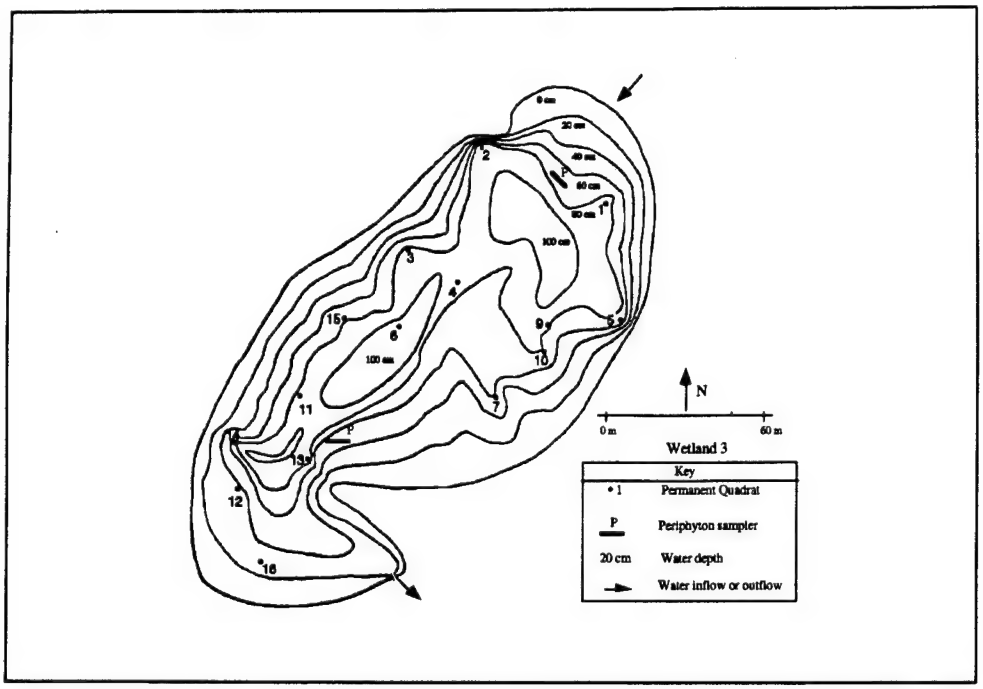

Figure 3. Experimental Wetland 3: Inflow and outflow locations, periphyton samplers, and permanent reference quadrats for vegetation and water column metabolism (Contours indicate average water depth in 20-cm intervals)

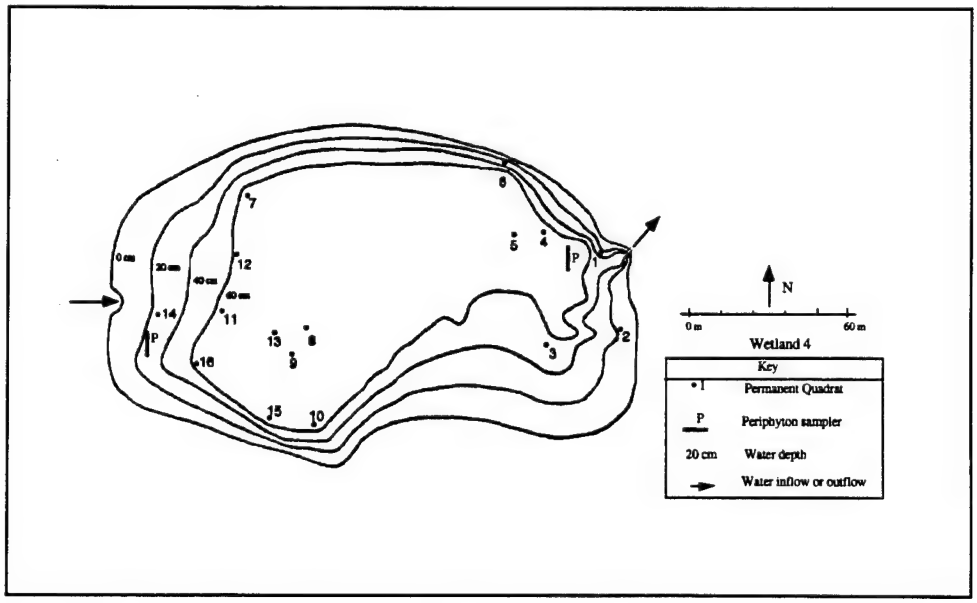

Figure 4. Experimental Wetland 4: Inflow and outflow locations, periphyton samplers, and permanent reference quadrats for vegetation and water column metabolism (Contours indicate average water depth in 20-cm intervals)

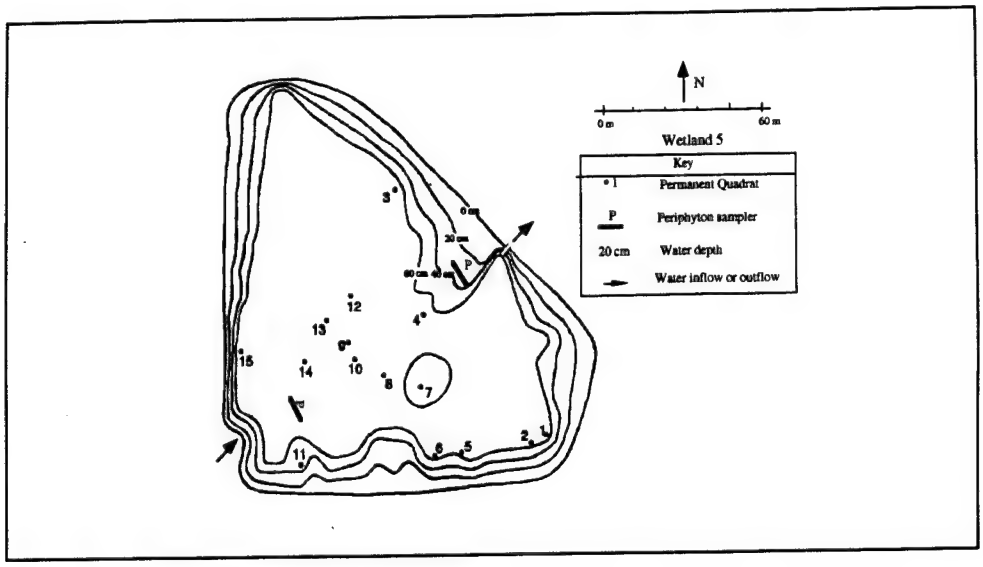

Figure 5. Experimental Wetland 5: Inflow and outflow locations, periphyton samplers, and permanent reference quadrats for vegetation and water column metabolism (Contours indicate average water depth in 20-cm intervals)

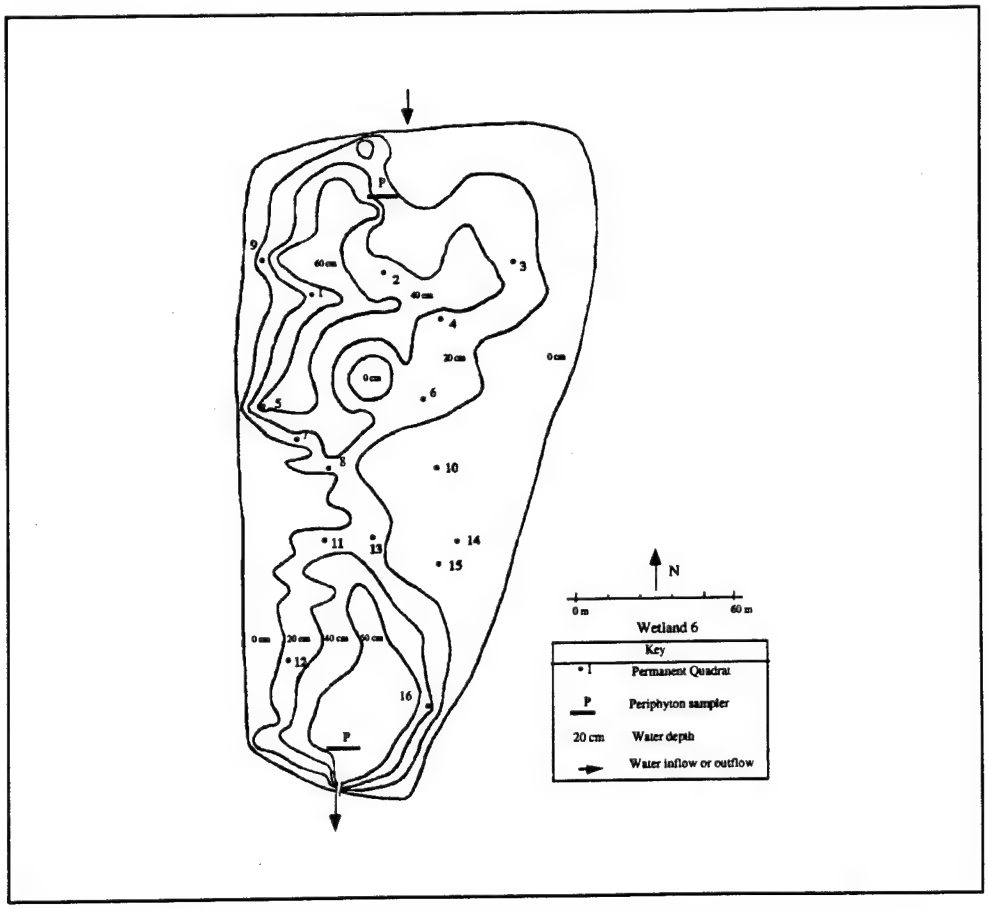

Figure 6. Experimental Wetland 6: Inflow and outflow locations, periphyton samplers, and permanent reference quadrats for vegetation and water column metabolism (Contours indicate average water depth in 20-cm intervals)

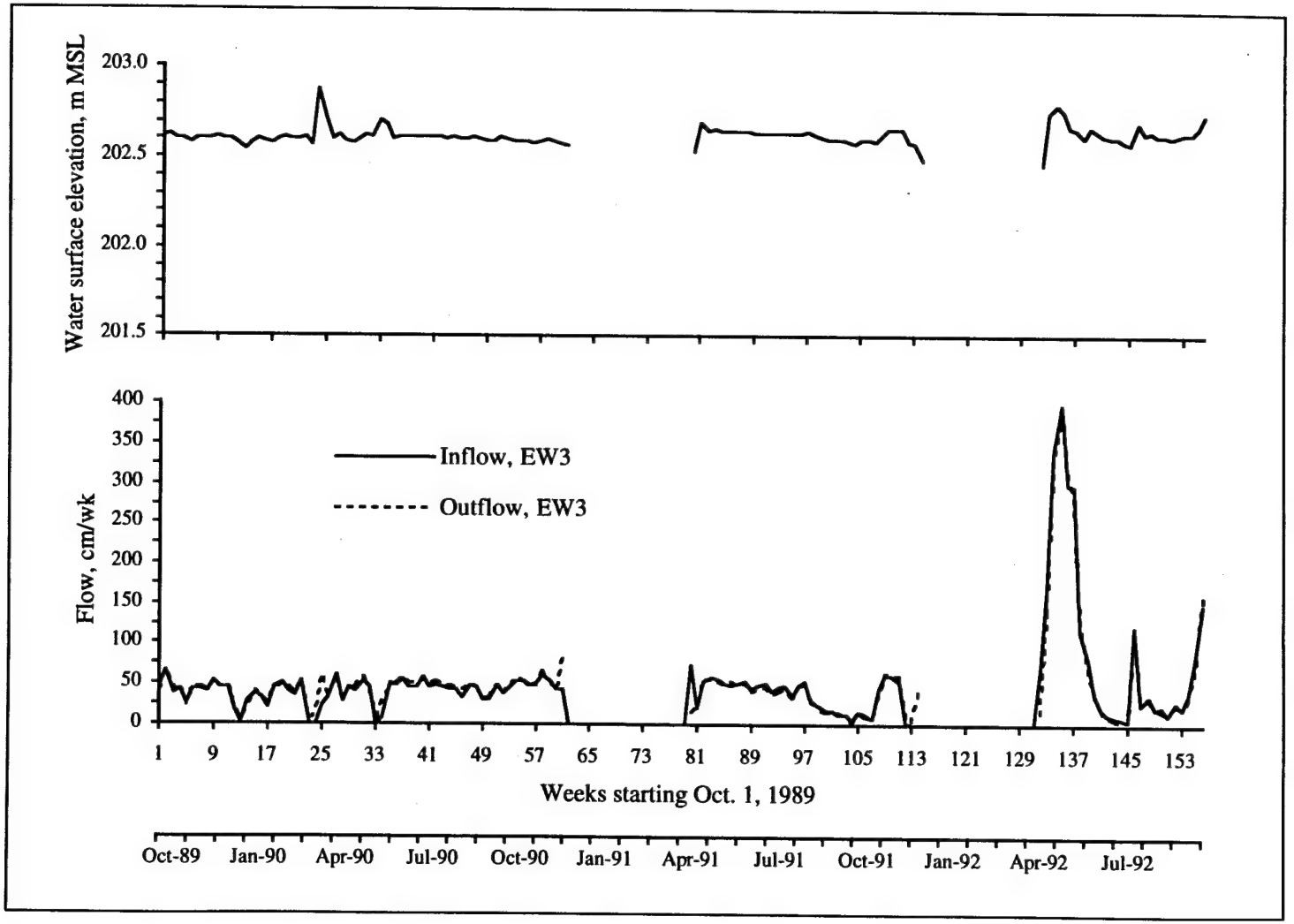

Figure 7. Hydrologic conditions of Experimental Wetland 3 (HFW) (October 1, 1989 - September 30, 1992) showing surface water level, pumped water inflow, and outflow at weir

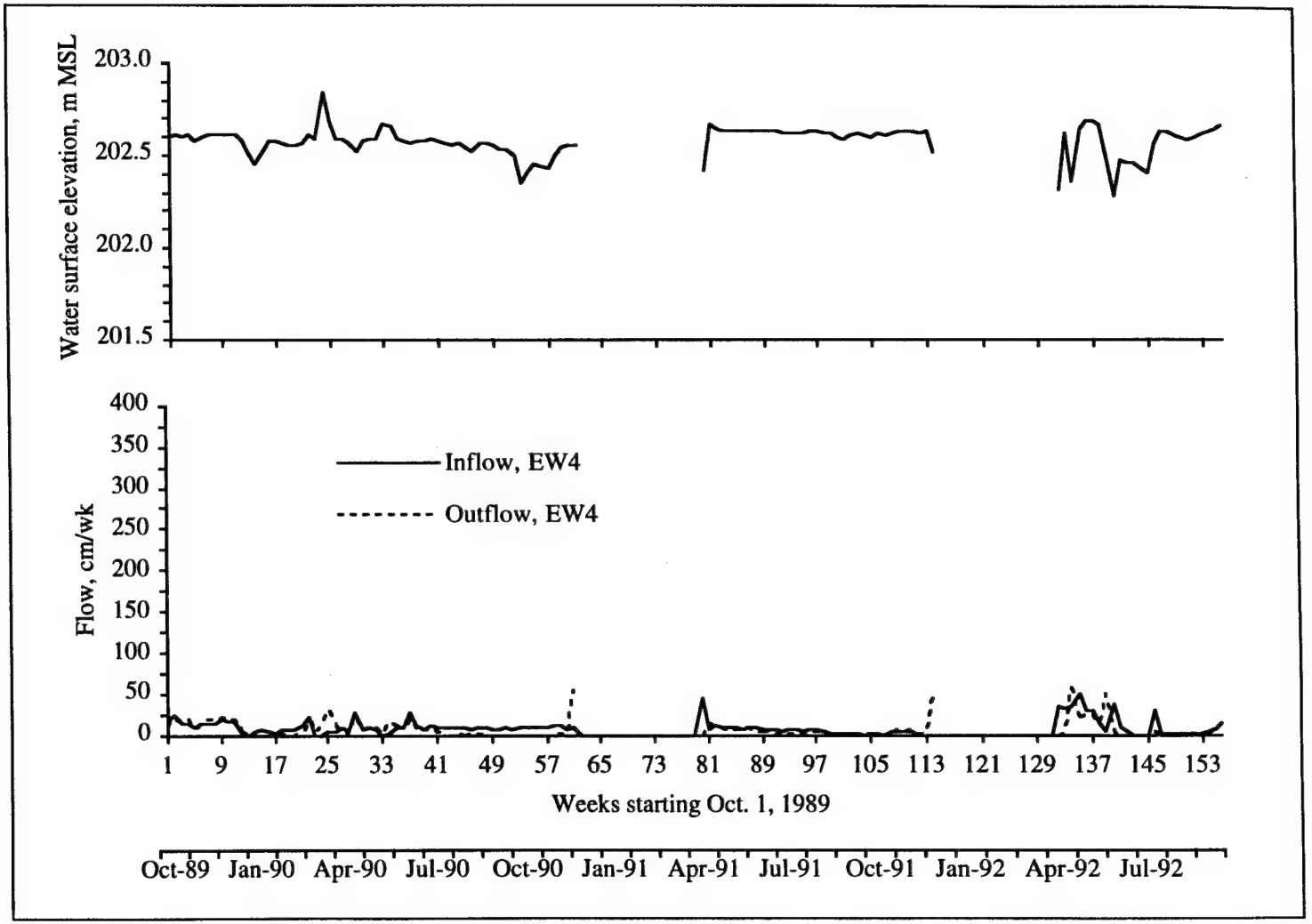

Figure 8. Hydrologic conditions of Experimental Wetland 4 (LFW) (October 1, 1989 - September 30, 1992) showing surface water level, pumped water inflow, and outflow at weir

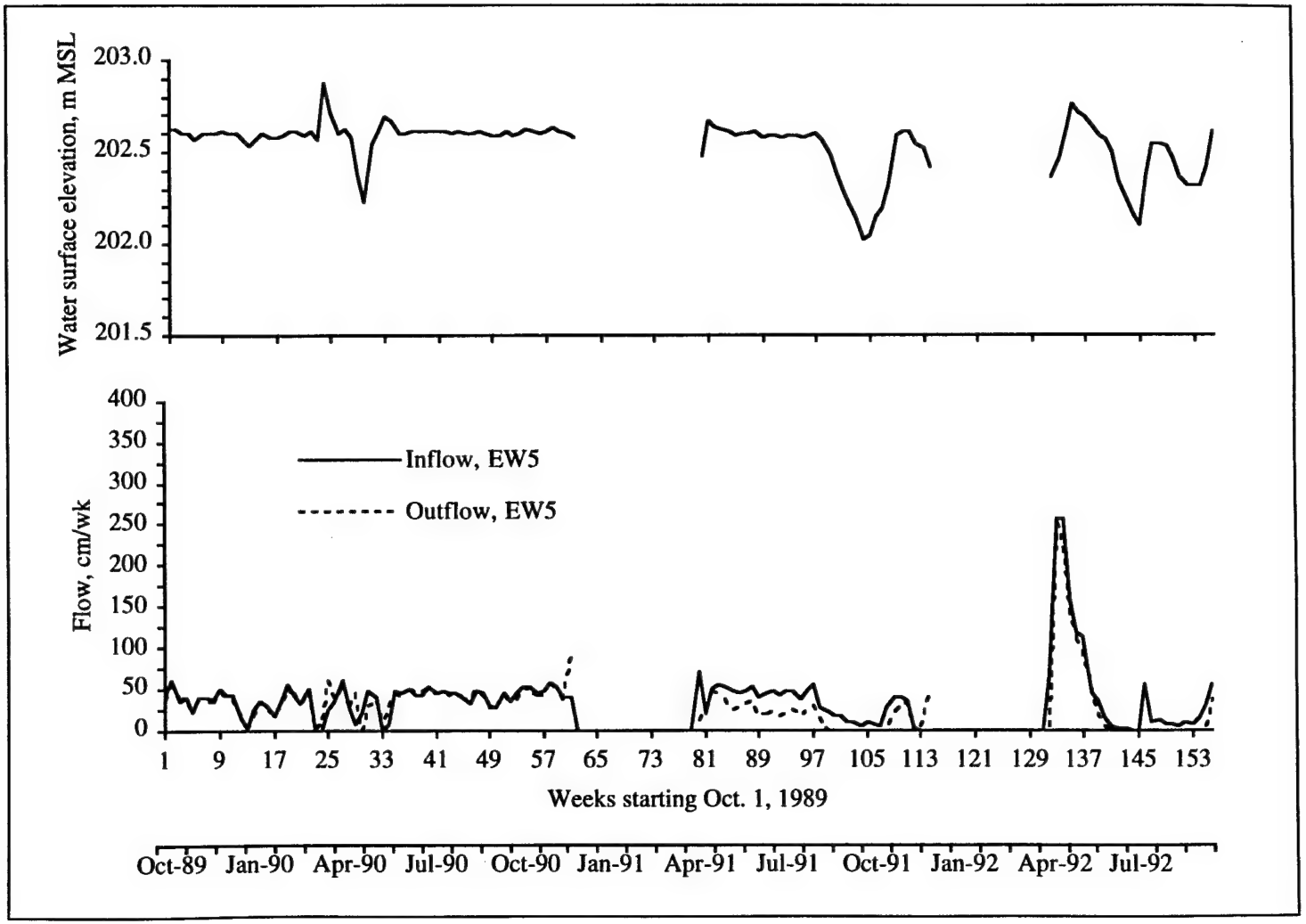

Figure 9. Hydrologic conditions of Experimental Wetland 5 (HFW) (October 1, 1989 - September 30, 1992) showing surface water level, pumped water inflow, and outflow at weir

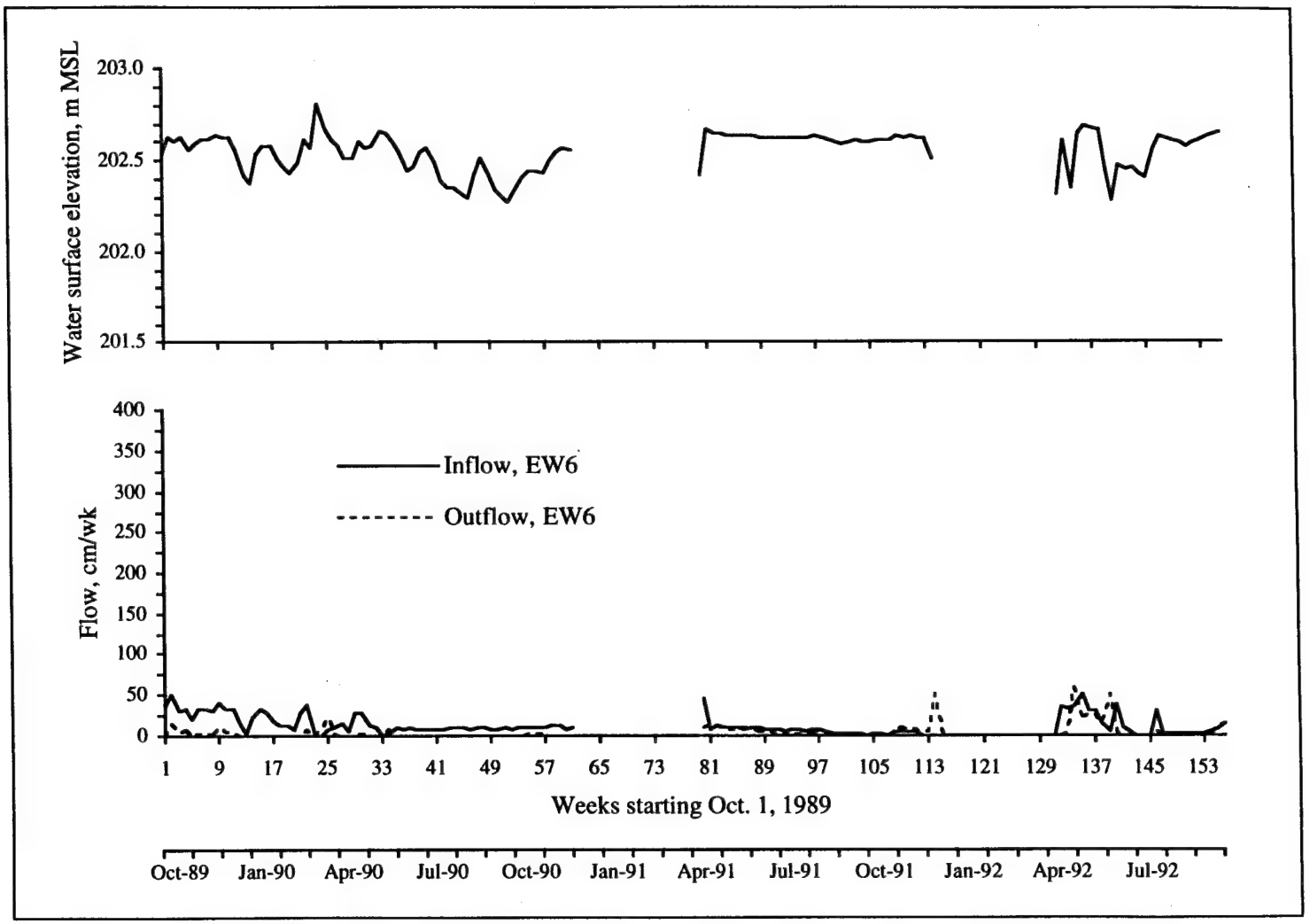

Figure 10. Hydrologic conditions of Experimental Wetland 6 (LFW) (October 1, 1989 - September 30, 1992) showing surface water level, pumped water inflow, and outflow at weir

2 Phosphorus Retention in Constructed Freshwater Marshes

Introduction

The concept is well established that natural freshwater wetlands often function as a sink for nutrients in high concentrations, especially for nitrogen and phosphorus (reviews by Nixon and Lee 1986; Johnston 1991). Nutrient removal in wetlands is facilitated by shallow water (which maximizes sediment-water exchange), waters of low velocity that allow sedimentation, often high productivity of vegetation, the presence of both aerobic and anaerobic sediments in proximity, and the accumulation of litter and eventually peat (Mitsch and Gosselink 1993). Less is known about constructed and restored wetlands subjected to near-ambient concentrations of nutrients. Because of the demonstrated success of natural and constructed wetlands in water quality improvement for high concentrations of nutrients (e.g., wastewater wetland studies such as Odum et al. 1977; Boyt, Bayley, and Zoltek 1977; Kadlec and Kadlec 1979; Dierberg and Brezonik 1983, 1984, 1985; Knight, McKim, and Kohl 1987; Brodrick, Cullen, and Maher 1988; Knight 1990), interest has been sparked in the construction of wetlands to improve water quality by controlling nonpoint source pollution (Olson 1992). Over half of the sources of water quality degradation in rivers and streams are nonpoint sources, predominantly agricultural runoff (Baker 1992).

Natural riparian zones, including their bottomland wetlands, retain solids and nutrients from the water column by slowing overland runoff (Peterjohn and Correll 1984; Johnston et al. 1984; Lowrance et al. 1984; Jacobs and Gilliam 1985), and many natural riparian wetlands that performed this function have been drained and bypassed. Further, rivers have been channelized, increasing stream water velocity and the load of suspended solids in rivers. Restoring wetlands adjacent to rivers might alleviate the burden of this problem. Mitsch, Dorge, and Wiemhoff (1979) determined that when an Illinois riparian cypress swamp was flooded, the swamp retained 10 times more phosphorus than it released back to the river (4.5 percent of the phosphorus that entered during a storm event). A floodplain swamp in North Carolina retained 62 to 66 percent of annual total

phosphorus inputs from a watershed that was 30 percent agricultural (Kuenzler et al. 1980). Peterjohn and Correll (1984), examining the role of a riparian forest in phosphorus retention in an agricultural watershed in Maryland, found that where farmers added 1.25 g $P \cdot m^{-2} \cdot year^{-1}$ to the cornfield, the riparian forest retained 0.3 g $P \cdot m^{-2} \cdot year^{-1}$. The phosphorus retention rate of a natural freshwater wetland adjacent to Lake Erie was also subject to changes because of changes in the loading rate. This wetland was estimated to retain from 17 to 52 percent of the incoming phosphorus, with the greatest retention occurring at times of high inflow and high lake levels (Mitsch and Reeder 1991).

Relatively few studies have documented the effectiveness of constructed, rather than natural, wetlands for retaining phosphorus in near-ambient (non-wastewater) conditions. In one of the few studies of this type, Craft, Broome, and Seneca (1989) used estuarine water to flood 0.1-ha transplanted diked marshes. They measured the effluent only 24 hr after flooding and found that phosphorus concentration was significantly reduced from inflow to outflow during each flooding event.

This chapter presents the findings of phosphorus retention over a 3-year period in constructed wetlands at the Des Plaines River site during which the wetlands were subjected to high- and low-flow conditions. Phosphorus sampling was undertaken to determine whether differences in phosphorus retention existed among the wetlands and between hydrologic regimes. Maps of the spatial distribution of phosphorus concentrations were developed to indicate whether phosphorus was short-circuited from inflow to outflow, if significant amounts of phosphorus disappeared near the inlet, or if phosphorus was preferentially adsorbed in areas of high-vegetation density or shallow depth. The fate of the phosphorus was estimated from measurements of wetland productivity in various communities (macrophytes, periphyton, and phytoplankton; see Chapters 3 - 5) and sedimentation. A simple linear model was developed to determine the effective uptake coefficient of the wetlands for different years and different hydrologic conditions.

Methods

Routine sampling

Wetlands Research, Inc., personnel took weekly water samples at the inflow to the wetlands and at all the outflows from October 1989 through September 1992 (sampling locations shown in Figures 3 - 6). Sampling indicated that all wetlands were receiving the same concentrations of phosphorus, so multiple inflow sampling was discontinued after 1990.

Spatial sampling

In 1991, 1-L surface samples were collected at each of the 15 to 16 permanent

quadrats (see Figures 3 - 6) within the wetlands on five sampling dates: May 16, June 24, July 22, August 20, and September 12. The quadrats were randomly placed throughout the wetlands and were originally established as part of a concurrent study on macrophyte vegetation (see Chapter 3). Samples were divided in two, and each half was preserved with 1 mL H_2SO_4. Samples were stored at 4 °C until analysis.

Intensive water quality sampling

ISCO 3700 automatic samplers loaded with 350-mL glass sample containers or 1,000-mL polyethylene containers were installed at the outflows of Wetlands 4 and 5. Samples from the inflows were collected by hand. Samples were collected at the outflows at 4-hr intervals throughout the day, and at the inflow daily during pumping. Every 2 days, the samples were removed from the automatic samplers. Samples were split into preserved (1 mL 18 M sulfuric acid/ L) and nonpreserved aliquots, labeled, packed in ice, and shipped via United Parcel Service to Columbus, OH, for analysis. Sampling logs were maintained for each sampler throughout the sampling interval, as were daily logs of weather and pumping schedules. Samples were collected for a trial period from August 12-22 in 1991 and continuously from May through September 1992.

Analysis

Phosphorus analyses of routine samples were performed at Abbott Laboratories, Chicago, in 1989-90 and then taken over by U.S. Environmental Protection Agency (USEPA) laboratories in Duluth, MN, in 1991 and 1992 using persulfate digestion and the ascorbic acid method. The USEPA laboratory allowed samples to settle before taking an aliquot for digestion and analysis. The detection level for phosphorus was 10 $\mu g\text{-}P \cdot L^{-1}$. Preserved nonfiltered samples from the spatial sampling of 1991 and intensive sampling of 1991 and 1992 were analyzed for total phosphorus at The Ohio State University using a persulfate digestion and the ascorbic acid method (American Public Health Association (APHA) et al. 1989) with a Lachat QuikChem IV flow injection analysis system, within the 10- to 1,000-$\mu g\text{-}P \cdot L^{-1}$ sensitivity range (Bloxham 1990). Samples were shaken just before aliquots were taken for digestion, thus including more suspended material in the sample than did the USEPA procedure. Soluble reactive phosphorus was determined on a portion of each sample that was filtered at 0.45 μm and measured with the ascorbic acid method (APHA et al. 1989).

Isopleth maps

Maps of phosphorus concentration were generated using the 1991 spatial data with a mapping software package called Surfer, which creates a regularly spaced grid from irregularly spaced data. The input includes the x and y coordinates of the sampling sites and the z value, which is the average concentration of the mapped parameter at each site. The interpolation between points is based on the

process of kriging, in which a linear relationship between data points is assumed, and the influence of one data point on another decreases with increasing distance (Golden Software, Inc. 1990). The maps were generated from average values of samples taken over the entire growing season of 1991 and are meant to show spatial trends rather than actual concentration values. Because of constant water movement, a map of concentrations at any given time would only be a snapshot and may not show trends as well as a map generated from average values.

Phosphorus fate estimates

Estimates reported in detail elsewhere of sedimentation (Fennessy, Brueske, and Mitsch, in press), macrophyte production (Chapter 3; Fennessy, Cronk, and Mitsch, in press), periphyton productivity (Chapter 4; Cronk and Mitsch in press), and water column productivity (Chapter 5; Cronk and Mitsch 1994) were used to make preliminary calculations of phosphorus processes. These are updates of previous estimates of phosphorus fate reported for 1 year by Mitsch (1992).

Modelling

The simulation package STELLA II was used on a Macintosh computer system to estimate phosphorus retention coefficients. Total annual outflow of phosphorus was used as a measure of data fit in calibration procedures, and the retention coefficient was used as the primary calibration variable. The model was simulated with a simple linear pathway, with a pathway affected by seasonal temperature fluctuations, and with an adjustment on the outflow to account for other flow leakages. The model was run with a time step of 0.1 week and a 2nd order Runge Kutta integration technique. This part of the study was partially supported by the USEPA, Region V.

Results

Phosphorus retention

Despite a variable pattern of routine sampling phosphorus concentrations in the river water delivered to the wetlands over the 3-year period, the average inflow concentration of total phosphorus was substantially greater than the average outflow concentration in all wetlands in all 3 years (T-test; $p < 0.01$; Table 2 and Figures 11 - 14), although very little difference was noted in the percent decrease in total phosphorus between the high-flow and low-flow wetlands for the first 2 years. For example, in the two mineral soil wetlands with the most complete database (Wetlands 4 and 5), the average outflow concentrations decreased by 63, 92, and 87 percent, respectively, for 1990, 1991, and 1992 in LFW 4 and by 62, 90, and 78 percent, respectively, in HFW 5 (Table 2). Third-year data from both routine and intensive sampling suggest, as had been originally hypothesized, that LFW 4 reduced the concentration of total

phosphorus more effectively than the HFW 3 and 5.

A similar pattern was observed for soluble reactive concentrations, which were generally 36 to 50 percent of the total phosphorus at the inflow, only 7 to 10 percent of the outflow total phosphorus concentrations in 1990, but 50 to 100 percent of the outflow total phosphorus in subsequent years. Soluble reactive phosphorus reduction in concentration ranged from 80 to 92 percent in LFW and 74 to 92 percent in HFW. In the LFW, retention of soluble reactive phosphorus decreased with each succeeding year.

Intensive sampling

The trial period for intensive sampling in 1991 indicated that the outflow total phosphorus concentrations were on average 88 and 90 percent lower than at the inflow in the LFW and HFW, respectively (Figure 15). Intensive sampling with 107 inflow samples and over 800 outflow samples in 1992 showed a decrease in phosphorus concentrations of 81 and 74 percent in the LFW and HFW, respectively (Figure 16). This compares well with 87 and 78 percent estimated from weekly samples analyzed by the USEPA laboratory from 30 inflow samples and 50 outflow samples during the same year. Outflow concentrations were measured (average ± standard error) as 34 ± 1 $\mu g\text{-}P \cdot L^{-1}$ in the LFW and 45 ± 1 $\mu g\text{-}P \cdot L^{-1}$ in the HFW in the intensive sampling and 16 ± 2 $\mu g\text{-}P \cdot L^{-1}$ and 27 ± 5 $\mu g\text{-}P \cdot L^{-1}$ in the LFW and HFW, respectively, in the weekly sampling (Table 2). Because samples analyzed in the intensive sampling procedure were shaken immediately prior to taking an aliquot for digestion whereas the weekly samples were allowed to settle, higher concentrations measured by the former method are expected. The slightly higher percent reductions of total phosphorus estimated from the weekly data compared with the intensive sampling data is reflected in the higher percent of phosphorus that is easily settleable from inflow (31 percent) to outflow (40 - 53 percent) in these wetlands (i.e., the percent of soluble and colloidal phosphorus decreases from inflow to outflow). This is not supported by investigation of the soluble reactive phosphorus (SRP)/total phosphorus (TP) ratios for 1992. SRP increased from 44 percent of TP in the inflow to 69 and 52 percent of TP in the LFW and HFW, respectively. By the third year, the HFW 5 discharged a greater concentration (and mass) of TP than did LFW 4, but a lower percent of its phosphorus was biologically available.

Spatial distribution of phosphorus

Soluble reactive phosphorus. Although both wetlands effectively removed SRP from the water column, the hydrologic regime does seem to play an important role in SRP distribution. The maps of SRP concentrations from 1991 indicate different patterns in the HFW and LFW (Figures 17 - 20). SRP isopleths reveal fairly homogeneous concentrations within LFW (Figures 18 and 20), but a gradual trend of decreasing concentrations from inflow to outflow in HFW (Figures 17 and 19). Approximately two-thirds of the SRP was removed from the water column very close to the inflow in LFW 4. SRP did not appear to be

preferentially adsorbed in areas of shallow depth, as there was no correlation between depth and SRP. Macrophyte density (calculated as percent shade cover; method in Cronk 1992) also appeared to have no relationship with SRP concentration. Means for all of the SRP measurements taken at the permanent reference quadrats indicated a clear difference between the two hydrologic conditions. The mean SRP concentration in HFW 5 (32 $\mu g\text{-}P \cdot L^{-1}$) was higher ($p < 0.10$) than that of LFW 4 (17 $\mu g\text{-}P \cdot L^{-1}$).

Total phosphorus. The maps of TP isopleths reveal the same trend as for SRP (Figures 21 - 24). In LFW, the average concentration is seen throughout the wetland; therefore TP is probably removed very near the inflow (Figure 22 and 24). In the HFW, total phosphorus is removed from the water column throughout the wetland and the TP concentration decreases gradually from inflow to outflow (Figures 21 and 23). Neither water depth nor macrophyte density correlated with TP concentrations.

Fate of phosphorus

A schematic of phosphorus fluxes and estimates of their values are illustrated in Figure 25 and Table 3. In 1990, the percent retention of the total mass of phosphorus was higher in the LFW than in the HFW. Higher percent retention in the LFW was expected since it receives a lower load. However, in 1991, HFW 5 showed very high percent retention (96 percent) of mass, and there was little difference between HFW and LFW in the percent removal of total phosphorus. In 1992, the LFW 4 once again retained a greater percentage of phosphorus than did HFW 3 (53 percent) or HFW 5 (78 percent). Overall, the LFW 4 retained 8 to 11 $mg\text{-}P/m^2\text{-}week$ while the HFW retained from 26 to 55 $mg\text{-}P/m^2\text{-}week$.

Sedimentation. Estimates of phosphorus retained in the wetlands are lower, sometimes by an order of magnitude, than that estimated as being retained by sediment traps. This suggests significant resuspension and a virtual spiraling of nutients from inflow to outflow. The data suggest a higher phosphorus sedimentation in the LFW in 1989-90 than in 1991, a situation that is probably unlikely and may be a measure of greater resuspension in that wetland.

Macrophyte uptake. Calculated uptake of phosphorus by emergent macrophytes is likewise higher than the phosphorus supplied by the inflow, but significant plant senescence leads to a great proportion of that uptake being released back into the water or sediments.

Periphyton and water column uptake. Uptake of phosphorus by periphyton, phytoplankton, and submersed aquatic vegetation (collectively called water column uptake) is estimated to be 4 to 6 $mg\text{-}P\ m^{-2}\text{-}week^{-1}$ (approximately 0.2 to 0.3 $g\text{-}P\ m^{-2}\text{-}year^{-1}$), well below the sedimentation rate or estimated emergent macrophyte uptake rate. However, much of the uptake by periphyton remains in insoluble form and falls to the sediments in the wetlands.

Model

A simple linear model (Figure 26) was developed and simulated in a companion project to this one to determine an overall decay coefficient of phosphorus in Wetlands 4 and 5 (mineral soils; one low flow, one high flow) for each of the 3 years for which data were available. The model is of the following form:

$$d[P]/dt = I - O - k[P]$$

where,

I = inflow mass of phosphorus, g-P/week (from field data)
O = outflow mass of phosphorus, g-P/week (from field data)
k = overall net retention coefficient, $week^{-1}$
$[P]$ = total phosphorus in water column, g-P

The model was run under simulations of a constant k, a k value influenced by seasonal temperatures (through a Q_{10} factor), and a k value influenced by temperature with an outflow corrected for seepage of some phosphorus to groundwater. A summary of the results is given in Table 4 where it is shown that the coefficient k varies from 11 to 244/year, depending on the simulation assumptions, year, and flow conditions. In general, the HFW had a higher k value than did the LFW. After correcting for temperature effects and leakage of water to groundwater, the k value is 16/year for the LFW and 60/year for the HFW. A higher k value represents a higher capacity for the wetland to retain phosphorus. A nonlinear relation exists where greater flow leads to greater phosphorus retention capacity. Apparently the phosphorus is being retained near the inflow of these wetlands, as seen by the spatial patterns described above, and the full capacity for phosphorus retention, at least of the LFW, has not yet been realized.

Discussion

High versus low flow

The results indicate that these wetlands effectively remove phosphorus from the incoming river water which has relatively low (100 to 200 $\mu g\text{-}P \cdot L^{-1}$) concentrations of phosphorus compared with phosphorus levels in most wastewater wetland studies. It was expected that the LFW would improve water quality more than the HFW because of the longer retention time. However, the high flow wetlands proved to be almost as efficient as the low flow wetlands in reducing phosphorus concentrations, and they were far more effective in removing mass of phosphorus than were the LFW (26 to 55 $mg\text{-}P/m^2\text{-}week$ versus 8 to 11 $mg\text{-}P/m^2\text{-}week$, not counting Wetland 6). This performance suggests that neither of the HFW was overburdened and that more phosphorus might be retained with even higher loadings. Using either calculation for phosphorus removal (percent of mass retained or percent decrease in

concentration), retention increased in both wetlands from 1990 to 1991, then decreased again from 1991 to 1992.

Spatial patterns

Significant spatial variation in water quality has been noted in a number of wetlands (Mitsch, Dorge, and Wiemhoff 1979; Chale 1982; Dierberg and Brezonik 1983; Robb 1989; Mitsch 1989). These studies show that sampling at several sites within a wetland, rather than solely at inflows and outflows, can help to illuminate removal mechanisms and to explain the biotic community's response to nutrient distribution (Pringle 1990). The isopleths of SRP and TP reveal an important difference between the hydrologic regimes. With greater phosphorus loading and slightly higher water velocity within the HFW, SRP and TP were distributed at higher concentrations throughout the wetland. This trend of decreasing concentration from inflow to outflow has been observed on a larger scale in other studies. For example, in two estuaries that lead to the Atlantic Ocean, phosphorus concentrations were consistently highest at the origin of the estuaries and lowest at the estuary mouths (Ward and Twilley 1986; Lebo 1990). The phosphorus distribution in the HFW at the Des Plaines site may stimulate the higher water column productivity that occurs throughout the HFW (Cronk and Mitsch 1994, in press).

In the LFW, phosphorus is distributed fairly evenly and at lower concentrations, because most of the phosphorus is removed from the water column very close to the inflow. Boyt, Bayley, and Zoltek (1977) also reported that the majority of the incoming TP to a Florida wastewater treatment wetland was removed from the water column very near the inflow. As sorption sites near the inflow of the LFW are taken up, the inflow areas may begin to show signs of "aging" in which available sorption sites for phosphorus become saturated and removal efficiencies decrease (Kadlec 1985). As a result, the sorption of phosphorus in the LFW may eventually occur farther and farther from the inflow.

Wetland performance over 3 years

The concentrations of phosphorus leaving Wetlands 3, 4, and 5 (Wetland 6 is excluded from this discussion because of excessive leakage) showed a general pattern of higher concentrations at higher flow conditions (Figure 27). When year-to-year variation is examined, the data show that the concentration of outflow phosphorus decreased in all three wetlands from the first to second years but increased during the third year. Figure 28 illustrates the percent of phosphorus mass retained in the three wetlands for 3 years. Again, considerable improvement is noted from the first year to the second, but percent mass retention decreased in the third year. It is clearly too early to tell if the wetlands are becoming saturated with phosphorus, as good percent retention (53 to 87 percent) is still occurring after 3 years of flow-through.

Table 2
Average Concentrations of Soluble Reactive Phosphorus and Total Phosphorus ± Standard Error (number of samples) at Inflow to the Wetlands and Outflow of each Wetland

Site	Soluble Reactive Phosphorus, μg P•L^{-1}			Total Phosphorus, μg P•L^{-1}			
	1990[a]	1991[b]	1992[b]	1990[a]	1991[b]	1992[b]	1992[c]
Inflow	40 ± 8 (19)	62 ± 6 (32)	54 ± 8 (30)	111 ± 12 (37)	132 ± 16 (32)	122 ± 14 (30)	176 ± 8 (107)
HFW 3 Outflow	5 ± 1 (21)	16 ± 2 (32)	14 ± 2 (29)	38 ± 5 (39)	27 ± 5 (25)	57 ± 5 (26)	-
LFW 4 Outflow	4 ± 1 (21)	8 ± 1 (31)	11 ± 1 (23)	40 ± 7 (35)	11 ± 2 (18)	16 ± 2 (24)	34 ± 1 (524)
HFW 5 Outflow	3 ± 1 (20)	15 ± 3 (30)	14 ± 2 (26)	42 ± 9 (37)	13 ± 3 (13)	27 ± 5 (26)	45 ± 1 (301)
LFW 6 Outflow	3 ± 1 (21)	9 ± 1 (17)	12 ± 2 (15)	25 ± 4 (37)	12 ± 1 (11)	27 ± 7 (13)	-

a Abbott Laboratories.
b USEPA Laboratories - Duluth.
c Ohio State University intensive water quality sampling

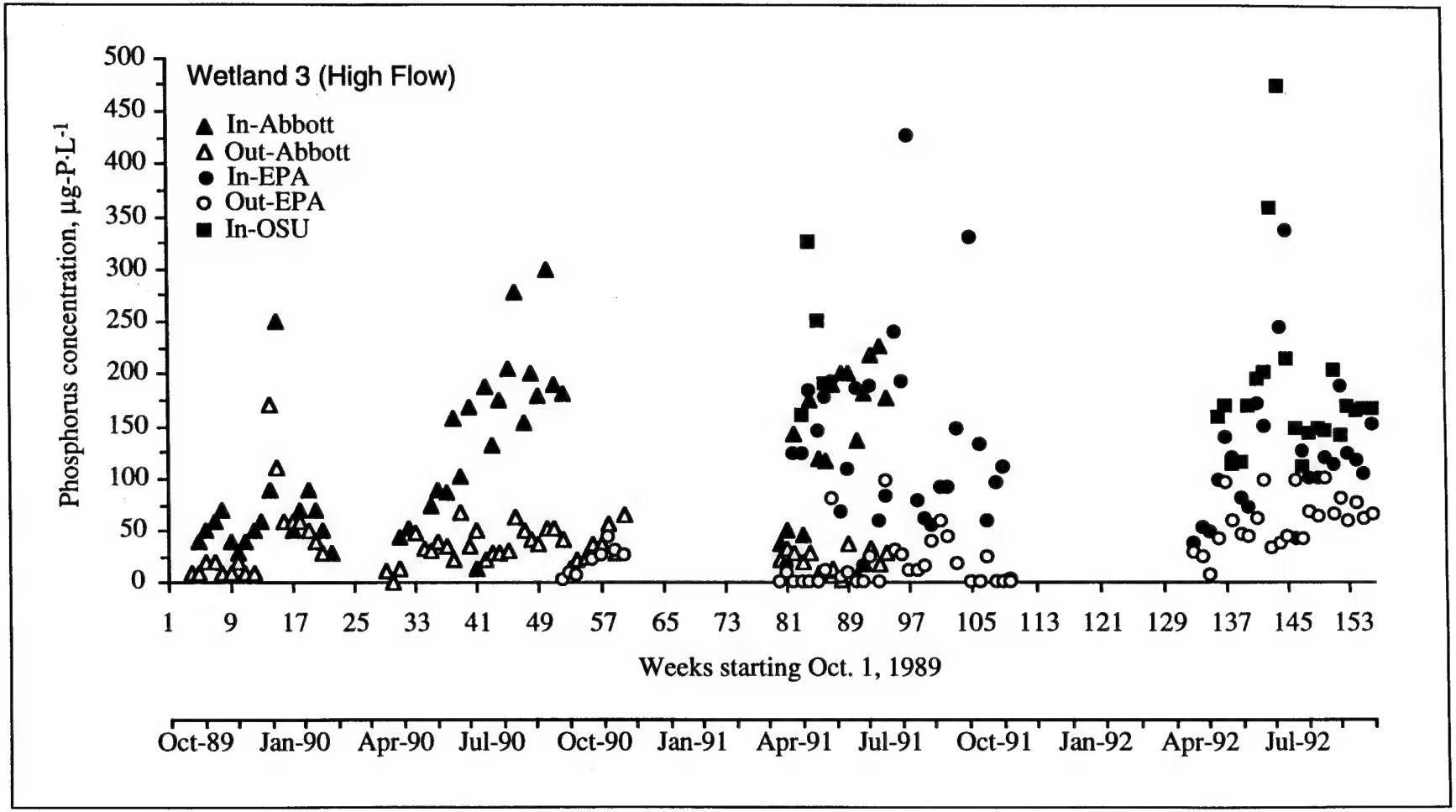

Figure 11. Total phosphorus concentrations of inflows and outflows of Experimental Wetland 3 (October 1989 - September 1992). Data provided by Abbott Laboratories, USEPA, and The Ohio State University

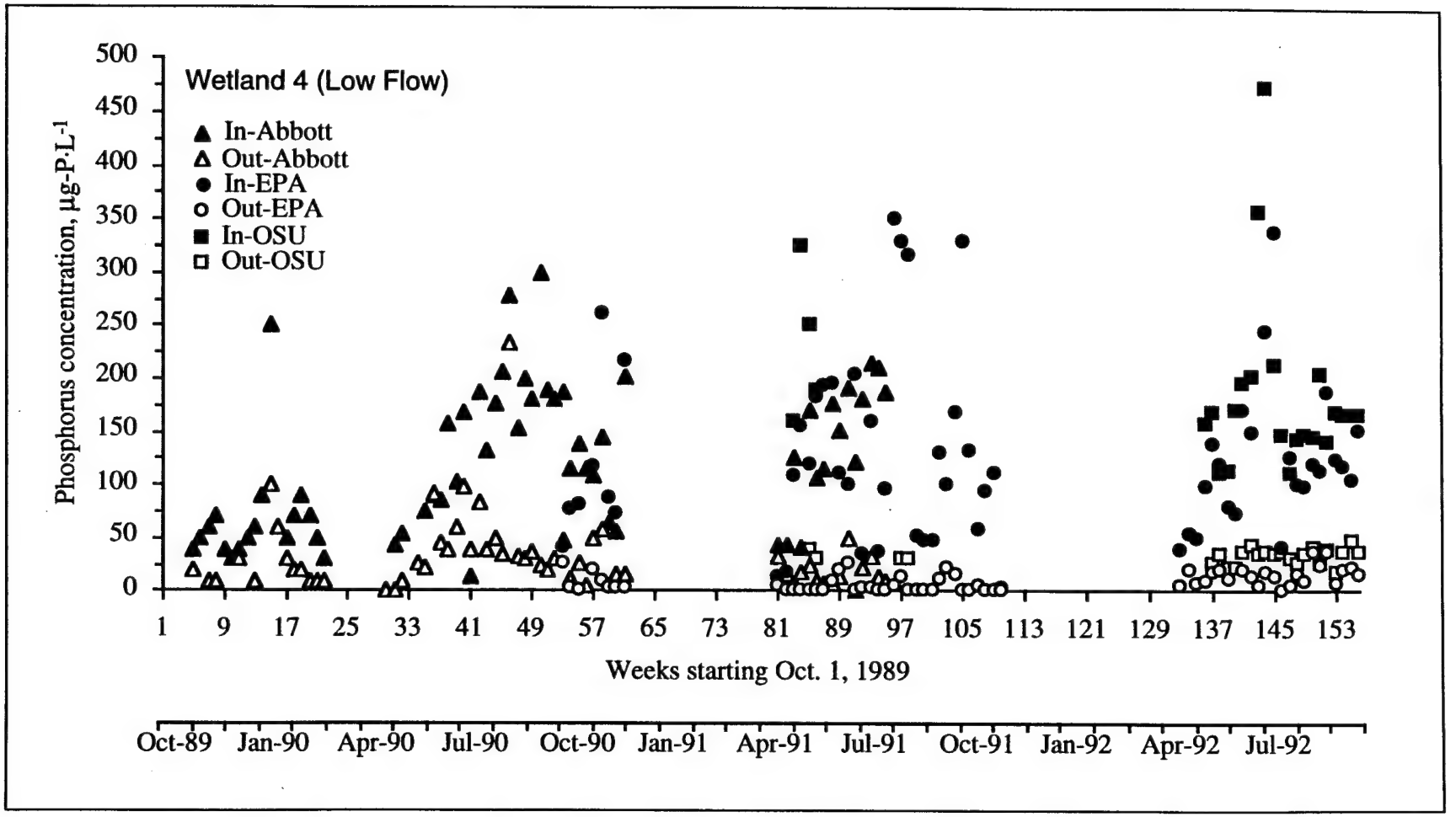

Figure 12. Total phosphorus concentrations of inflows and outflows of Experimental Wetland 4 (October 1989 - September 1992). Data are those provided by Abbott Laboratories, USEPA, and The Ohio State University

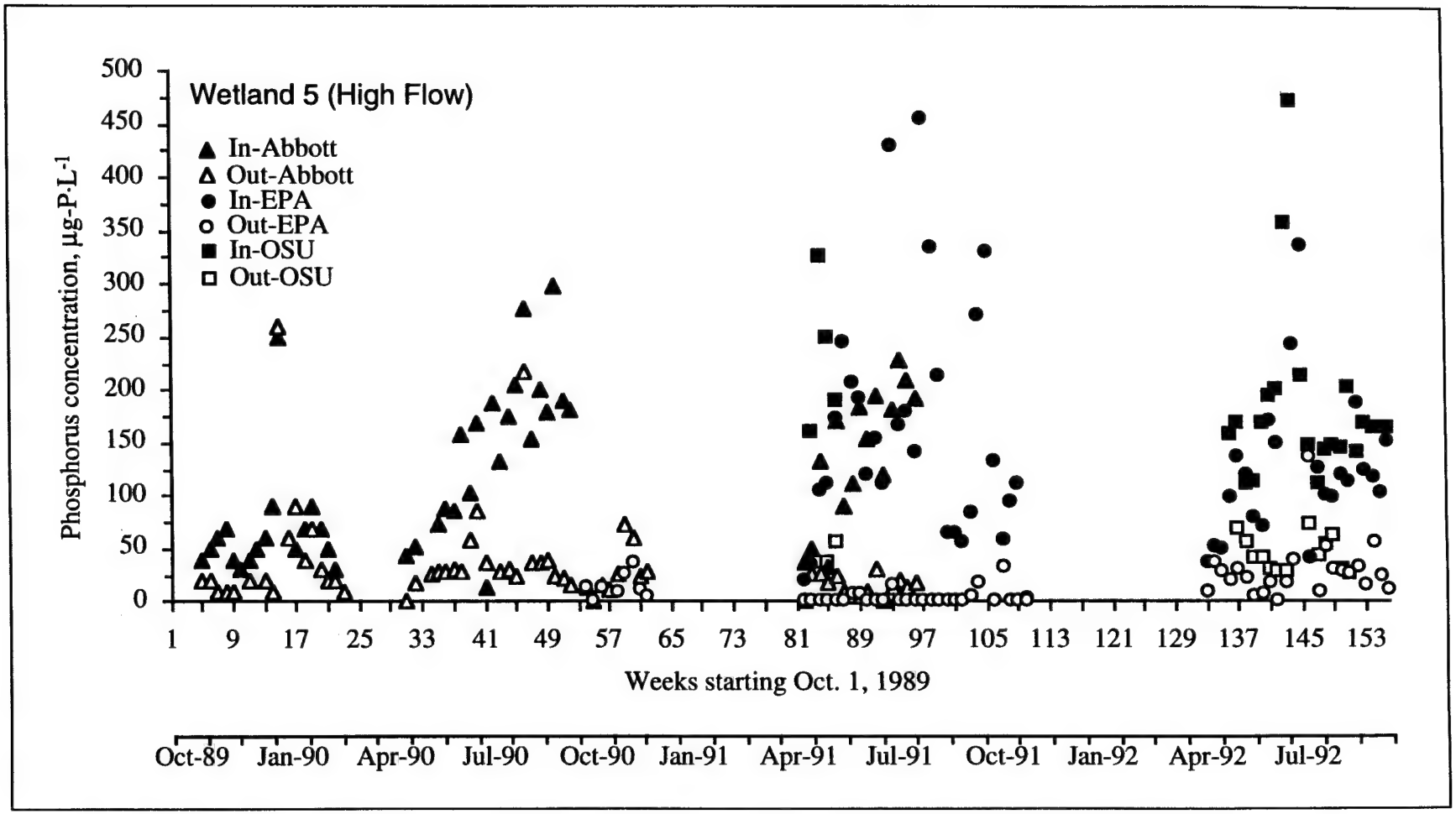

Figure 13. Total phosphorus concentrations of inflows and outflows of Experimental Wetland 5 (October 1989 - September 1992). Data provided by Abbott Laboratories, USEPA, and The Ohio State University

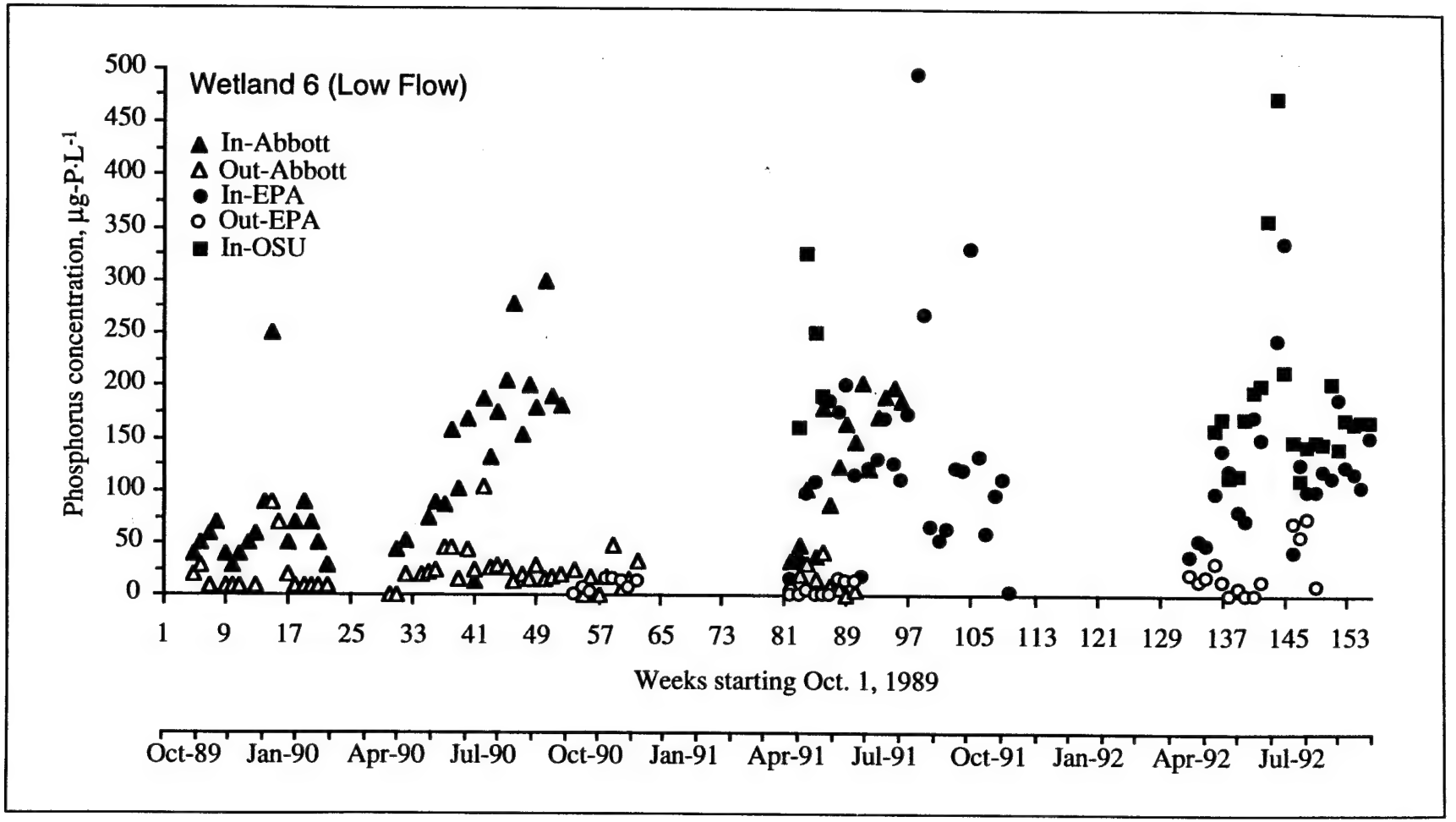

Figure 14. Total phosphorus concentrations of inflows and outflows of Experimental Wetland 6 (October 1989 through September 1992). Data are those provided by Abbott Labs, USEPA, and The Ohio State University

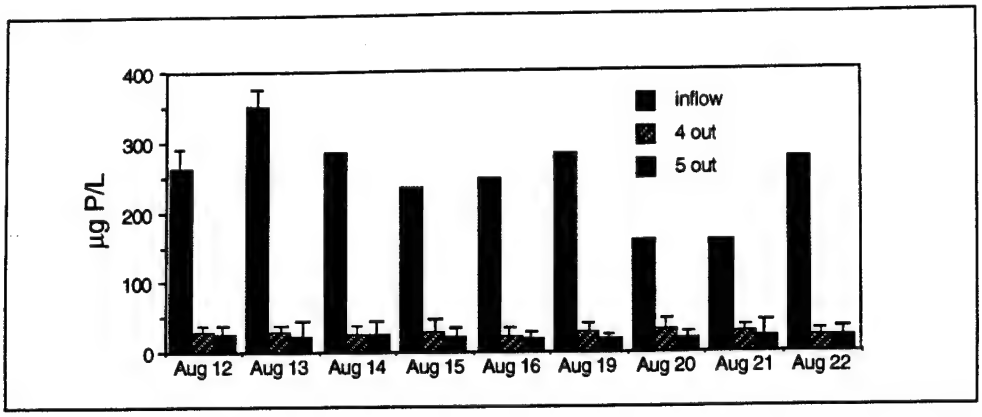

Figure 15. Total phosphorus concentrations at inflow versus outflow in Wetlands 4 and 5 for intensive sampling in August 1991. Each outflow data point is the average of six samples. Bars indicate standard error

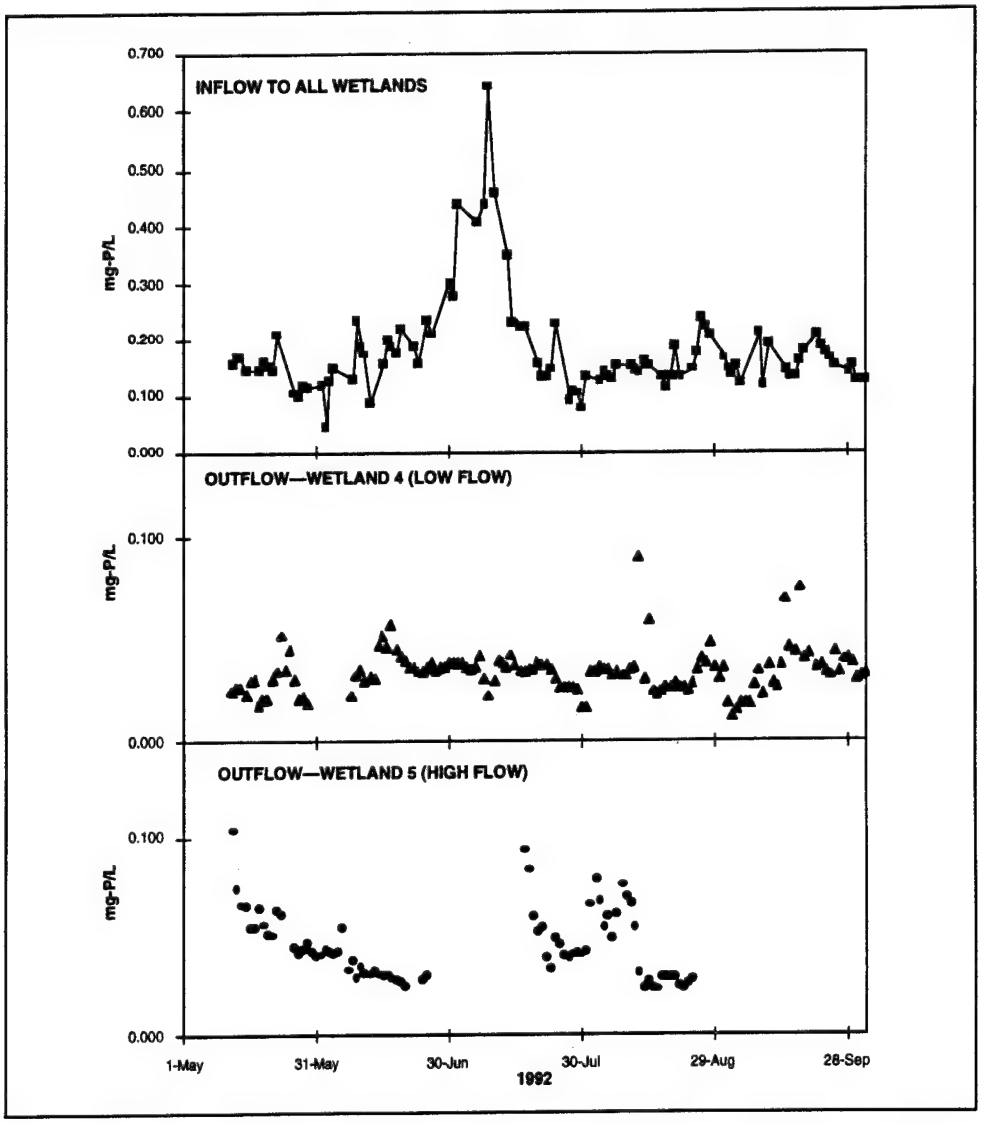

Figure 16. Results of intensive water quality sampling in 1992 with inflow and outflow from Wetlands 4 and 5. Most data points represent average of six samples taken at 4-hr intervals

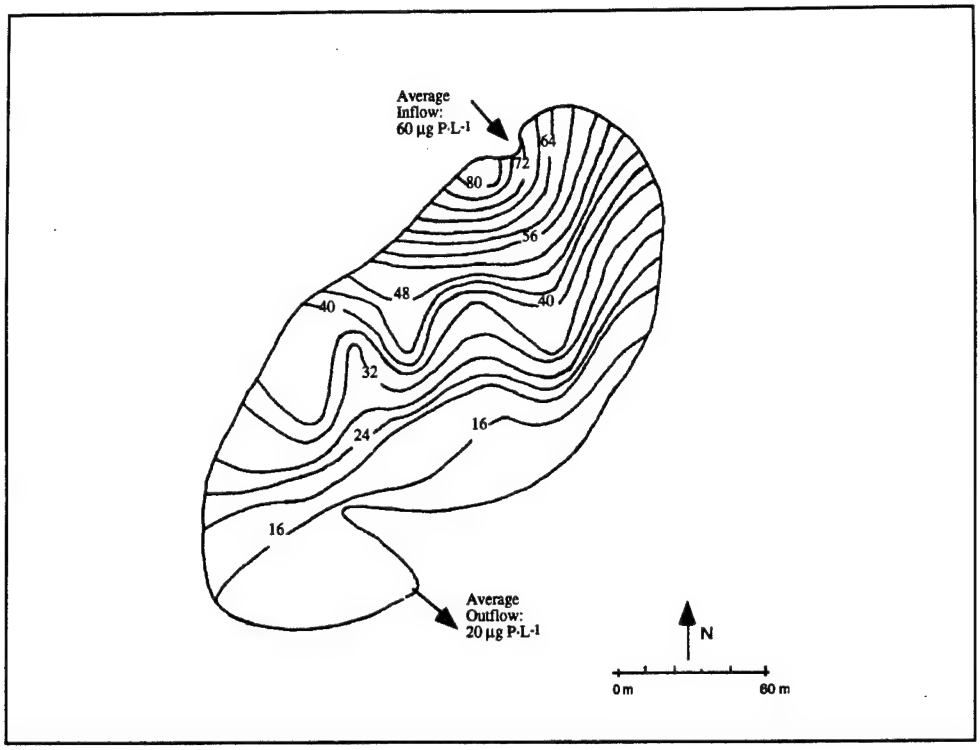

Figure 17. High-Flow Wetland 3: Isopleths of average soluble reactive phosphorus concentrations in increments of 4 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at the inflow and outflow arrows

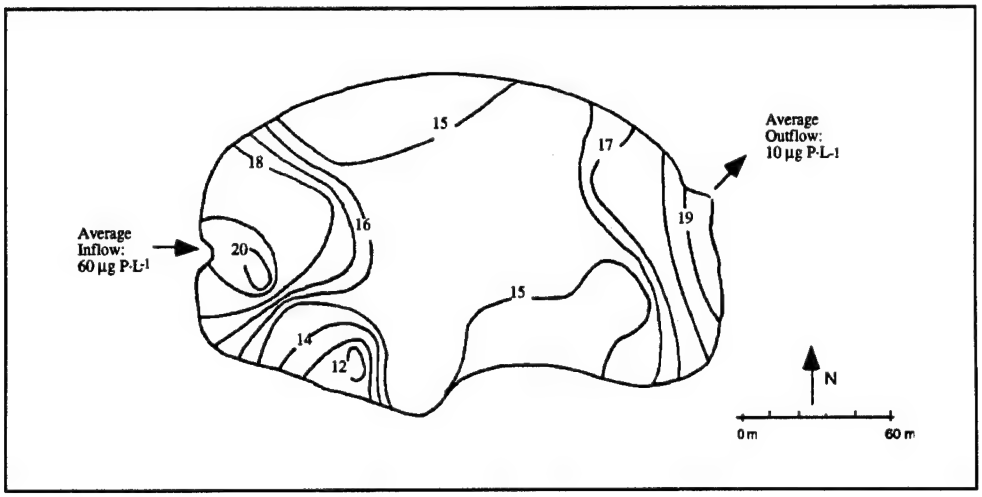

Figure 18. Low-Flow Wetland 4: Isopleths of average soluble reactive phosphorus concentrations in increments of 1 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at inflow and outflow arrows

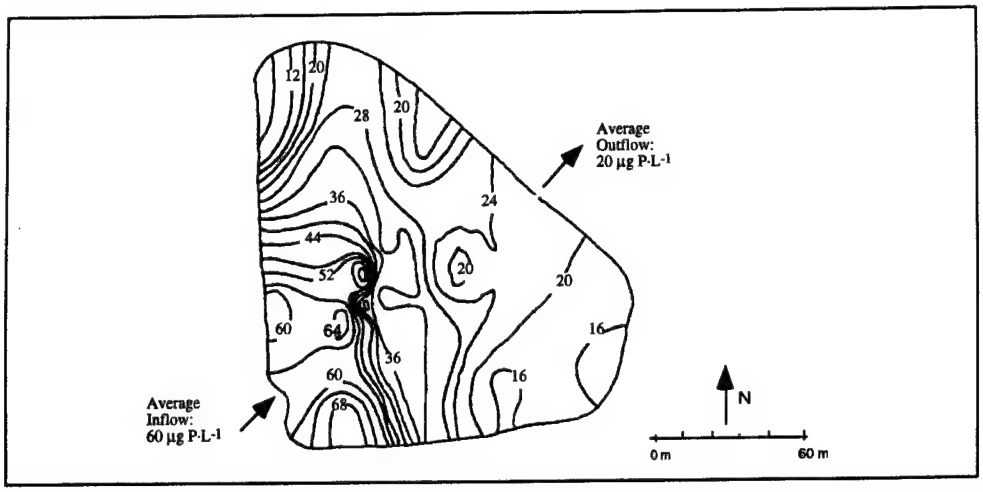

Figure 19. High-Flow Wetland 5: Isopleths of average soluble reactive phosphorus concentrations in increments of 4 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at inflow and outflow arrows

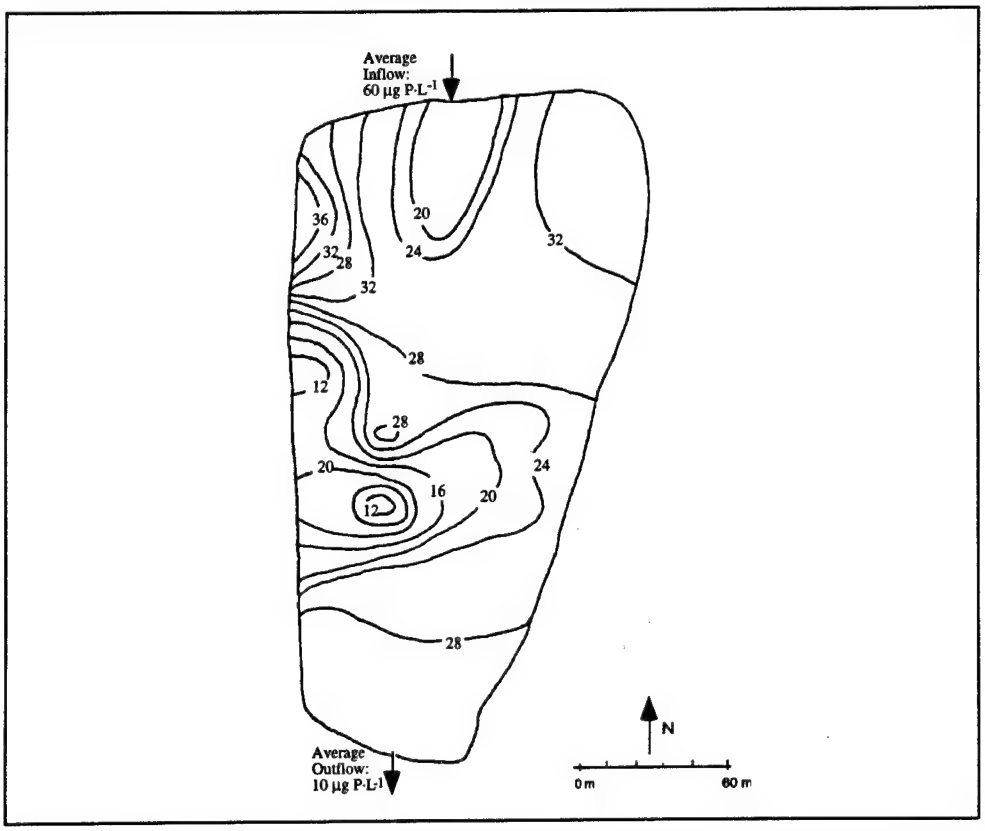

Figure 20. Low-Flow Wetland 6: Isopleths of average soluble reactive phosphorus concentrations in increments of 4 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at inflow and outflow arrows

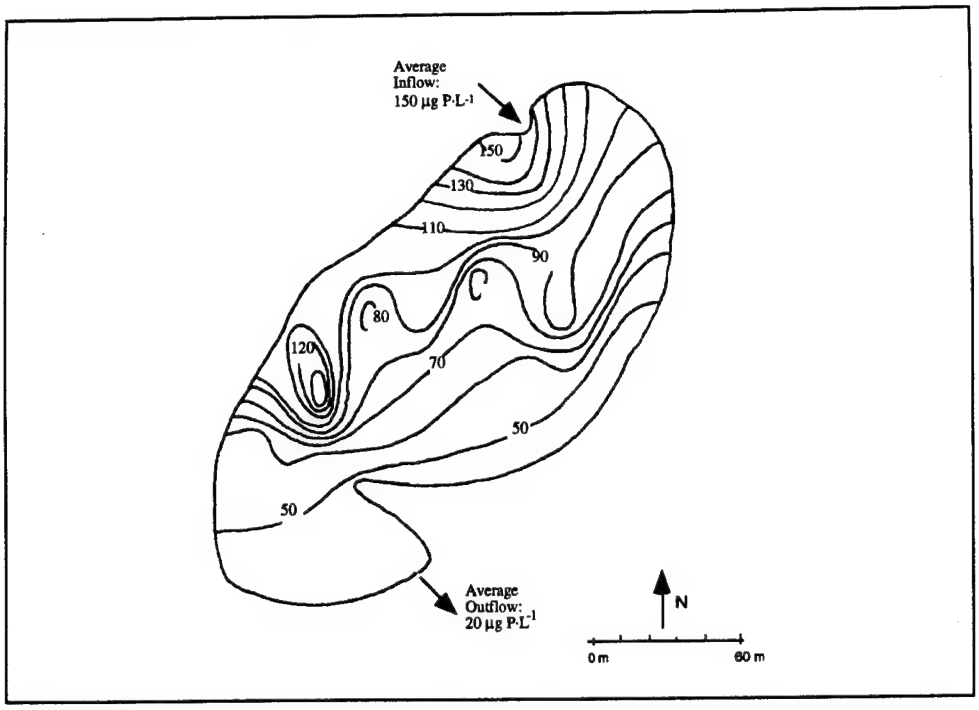

Figure 21. High-Flow Wetland 3: Isopleths of average total phosphorus concentrations in increments of 10 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at inflow and outflow arrows

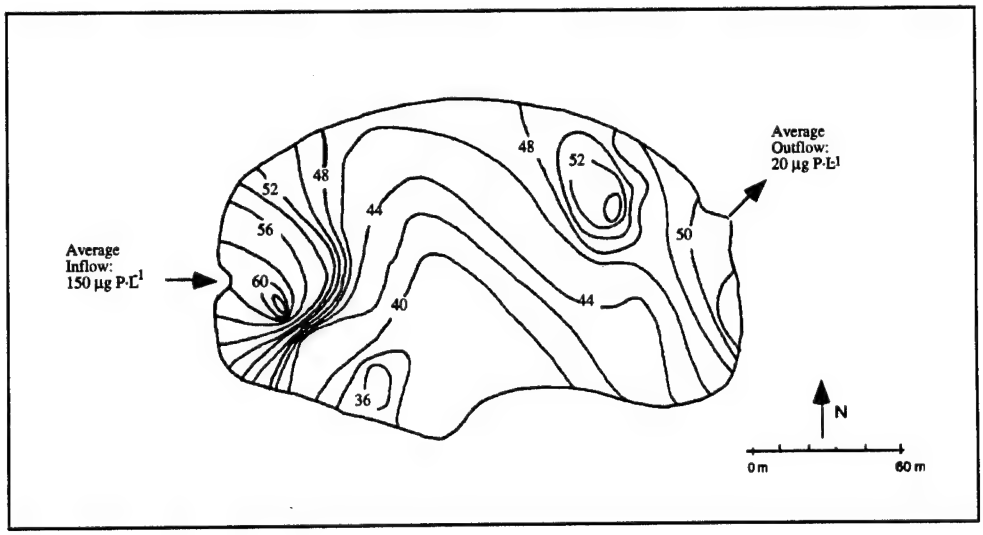

Figure 22. Low-Flow Wetland 4: Isopleths of average total phosphorus concentrations in increments of 2 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at inflow and outflow arrows

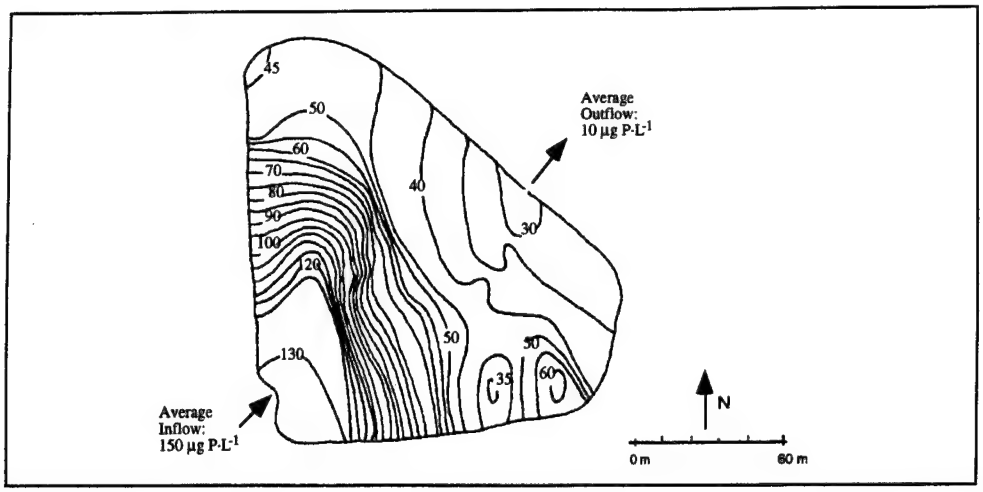

Figure 23. High-Flow Wetland 5: Isopleths of average total phosphorus concentrations in increments of 5 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at inflow and outflow arrows

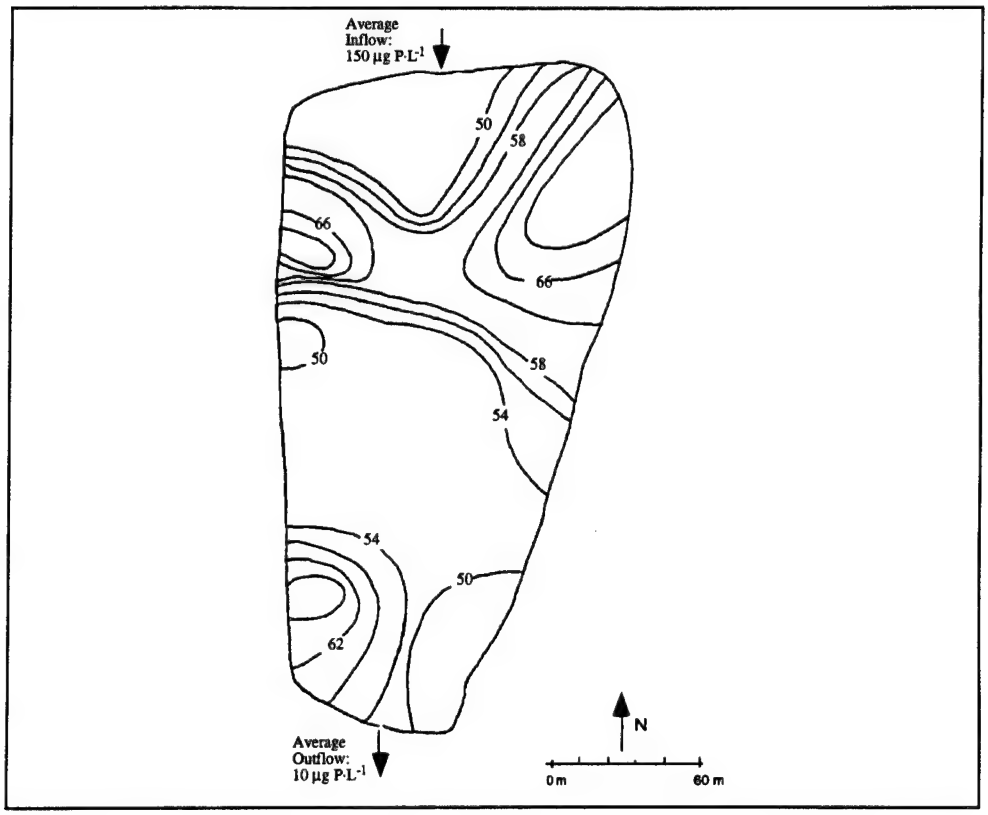

Figure 24. Low-Flow Wetland 6: Isopleths of average total phosphorus concentrations in increments of 4 µg $P \cdot L^{-1}$. Averages are based on monthly samples taken at 16 permanent reference quadrats from May - September 1991. Average inflow and outflow values for same sampling dates are shown at inflow and outflow arrows

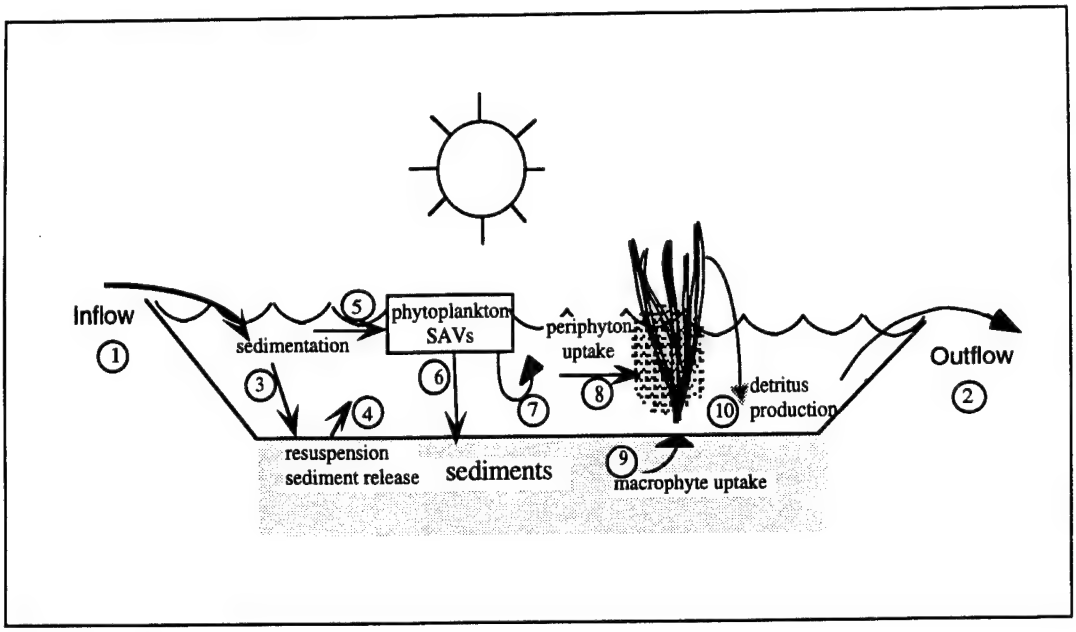

Figure 25. Major phosphorus fluxes in experimental wetlands

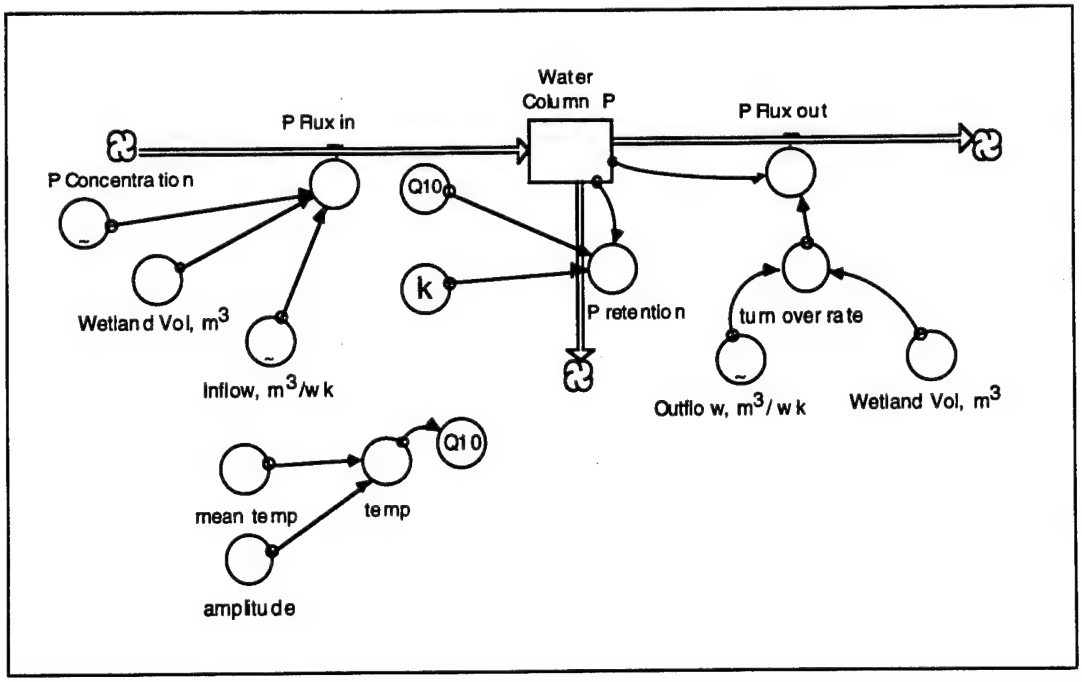

Figure 26. Vollenweider-type inflow-outflow phosphorus model used to estimate phosphorus retention coefficient

Table 3
Estimates of Phosphorus Fluxes in Experimental Wetlands

Year	1 Inflow	2 Outflow	1 - 2 Retention	% Retention	3+6+10 Gross Sedimentation[a]	9 Macrophyte Uptake[b]	5+8 Water Column Uptake[c]	8 Periphyton Uptake[d]	5 Phyto/SAV Uptake[e]
HFW 3									
1990	44.1	16.3	27.8	63	252	51	5	—	
1991	58.7	8.3	50.4	86	233	25	5	—	—
1992	77.2	36.5	40.7	53	—	20	5	—	—
LFW 4									
1990	10.5	2	8.5	81	153	77	4	—	—
1991	9.9	0.2	9.7	98	67	91	4	2	2
1992	9.0	1.2	7.8	87	—	102	4	—	—
1992 (intensive)	12.3	1.6	10.7	87	—	—	—	—	—
HFW 5									
1990	41.9	15.5	26.4	63	113	71	6	—	—
1991	57.3	2.3	55.0	96	141	18	5	3	2
1992	35.4	7.9	27.5	78	—	119	2	—	—
1992 (intensive)	32.4	5.4	27.0	83	—	—	—	—	—
LFW 6									
1990	13.7	0.1	13.6	99	117	12	6	—	
1991	11.4	0.2	11.2	99	—	12	3		
1992	40.0	6.8	33.2	83	—	41	—	—	—

Note: Values except percent retention and ratios are mg-P $m^{-2}week^{-1}$. Numbers in header refer to pathways in Figure 25.

[a] From sedimentation measurements and analysis of sediment trap contents for total P from 1989-90.

[b] Macrophyte NPP (above + below ground) × average P tissue concentration (for *Typha* sp.; 2.1 mg P/g dry weight), averaged over 52 weeks.

[c] Assumes 0.8 mgP/g dry weight measured for *Cladophora* and similar species × estimated water column NPP, averaged over 52 weeks.

[d] Estimated from periphyton productivity estimates near inflow and outflow, averaged.

[e] Calculated by subtracting estimated periphyton uptake from total water column uptake.

Table 4
Results of Simulation Calibration for Low- and High-Flow Mineral Soil Wetlands at Des Plaines River Wetland Site

Simulation, Wetland, and Year	Total P Outflow, gP/m^2 [a]			*k*	
	Field	Model	% error	$week^{-1}$	$year^{-1}$
Static k					
LFW 4[b]					
1990	0.03	0.03	-2.20	0.35	18
1991	0.02	0.02	-1.99	0.8	42
1992	0.03	0.04	-4.05	0.7	36
average				**0.62**	**32**
HFW 5[c]					
1990	0.44	0.44	-1.34	1.7	88
1991	0.18	0.18	0.23	4.7	244
1992	0.10	0.11	-0.51	2.1	109
average				**2.83**	**147**
Static *k* Plus Temperature Effect					
LFW 4					
1990	0.03	0.03	1.68	0.25	13
1991	0.02	0.02	1.99	0.39	20
1992	0.03	0.03	1.26	0.3	16
average				**0.31**	**16**
HFW 5					
1990	0.44	0.44	0.01	0.6	35
1991	0.18	0.18	-0.32	1.9	99
1992	0.10	0.11	-1.58	0.9	47
average				**1.16**	**60**
Static *k* Plus Temperature Effect Plus Inflow = Outflow					
LFW 4					
1990	0.03	0.03	-0.1	0.22	11
1991	0.02	0.02	-1.2	0.39	20
1992	0.03	0.03	0.3	0.35	18
average				**0.32**	**17**
HFW 5					
1990	0.44	0.43	0.3	0.65	34
1991	0.18	0.18	-0.2	1.9	99
1992	0.10	0.10	0.3	0.85	44
average				**1.13**	**59**

Note: Retention coefficient, *k*, is calculated by matching model outflow to measured outflow.

[a] Simulation for weeks 17 - 39 (23-week period.)
[b] LFW 4 = 2.34 ha.
[c] HFW 5 = 1.87 ha.

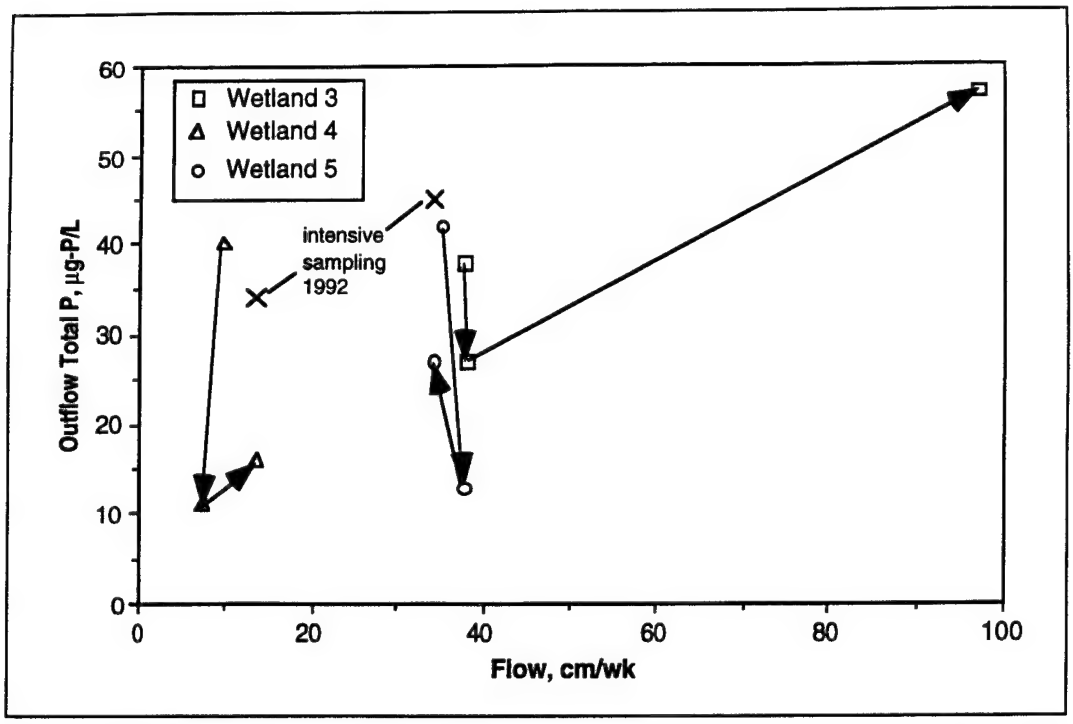

Figure 27. Annual average outflow phosphorus concentration versus average inflow flow rate for three experimental wetlands at Des Plaines River Wetland site for 3 years. Outflow concentrations estimated from intensive sampling in 1992 are also shown. Arrows indicate trends from water year 1990 to 1991 to 1992 for each wetland

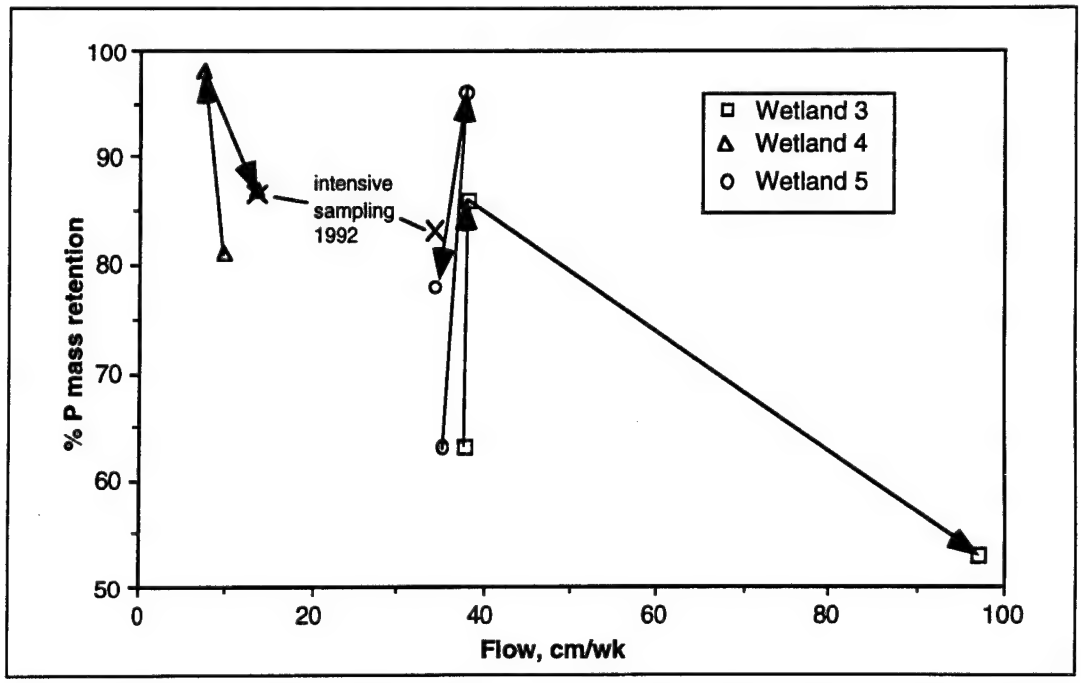

Figure 28. Annual average phosphorus retention (percent mass) versus average inflow flow rate for three experimental wetlands at Des Plaines River Wetland site for 3 years. Outflow retention estimated from intensive sampling in 1992 is also shown. Arrows indicate trends from water year 1990 to 1991 to 1992 for each wetland

3 Five Years of Macrophyte Community Development in Constructed Freshwater Marshes[1]

Introduction

This report is an update of the development of the wetland plant community in newly constructed freshwater wetlands at the Des Plaines River Wetland Research site through 1992. Species composition and successional changes have been monitored in four experimental wetlands for 5 years through this and previously funded projects (see, e.g., Fennessy, Cronk, and Mitsch 1992, 1994). The purpose was to determine the successional patterns of newly constructed wetlands and whether different flow rates into the wetlands would affect macrophyte productivity. It was hypothesized that forcing functions to the system (water, nutrient, and sediment inflows) would eventually lead to a difference in some aspects of the plant community structure and that higher flow rates would lead to higher rates of net aboveground productivity.

Methods

Data collection on the macrophyte communities has spanned five growing seasons from 1988-1992. Methodology varied slightly because of different site conditions each year. Details of methodologies for 1988 - 1991 are reported elsewhere (Fennessy, Cronk, and Mitsch 1992, 1994) and are only summarized here.

1988: Preflooding conditions

As experimental conditions had not been established at the site, vegetation sampling was restricted to a baseline census of the vegetation in each basin

[1] This chapter was originally written by M.S. Fennessy and was updated with 1991 and 1992 data.

during late July - early August. Plants in the excavated wetland basins were sampled at 25-m intervals along transects established in each basin. At each location, a 0.5-m^2 quadrat was surveyed to identify and determine the frequency of all species present. All individuals were identified to species and the number of stems counted for each (Moore and Chapman 1986). Six to eight transects were sampled per basin for a total of approximately 70 quadrats in each.

1989: First year of flooding

A census of the species present in the wetlands was conducted using the same method as in 1988. In addition, peak biomass production in each wetland was estimated by harvesting seven to ten 0.5-m^2 quadrats in each basin. Each wetland was stratified into three equal areas (nearest inflow, middle, nearest outflow), and two to four quadrats were randomly selected within each zone (Moore and Chapman 1986). From July 20 - 27, quadrats were marked and all plants within them were cut at the soil surface. In the laboratory, plants were separated by species, rinsed, and oven dried to a constant weight at 105 °C.

1990-1991: Constant flow conditions

Because 1990 was the first full growing season under experimental conditions, biomass measurements were expanded. Each wetland was stratified into three equal areas parallel to the direction of flow and five to six 4.0-m^2 randomly selected plots were permanently established in each area. This resulted in a total of 15 to 16 permanent reference quadrats per wetland (see Figures 3 - 6). Within each 4-m^2 plot, a 0.5-m^2 subsection was marked off, and all intensive measurements were conducted in this area. Plants in the 0.5-m^2 subplots were monitored in situ on a monthly basis from May to August. Each sampling period lasted 4 days or less. Biomass was estimated by measuring leaf lengths for all emergent species and leaf diameter for floating-leaved species. Stems of submersed species were measured for length. On each sampling date, plants from outside the reference quadrats representing the range of sizes found inside the quadrats were measured, harvested, and oven dried; these data were used to construct linear regressions of leaf length versus leaf weight.

1992: Pulsing flow conditions

A survey of the species present in each wetland was performed on a monthly basis from May to October. In addition, peak biomass production was estimated by harvesting several 0.5-m^2 quadrats in each wetland from August 6 - 9 ($n = 5$ in Wetland 3, $n = 12$ in Wetland 4, $n = 7$ in Wetland 5, and $n = 12$ in Wetland 6). Plant height was measured in the field and plant species were harvested within each quadrat. In the laboratory, plants were separated by species, rinsed, and oven dried to a constant weight at 105 °C.

Similarity index

Species composition was recorded each year using the same methodology as in 1988. To measure the similarity of species composition in the four experimental wetlands, the number of species common to the four wetlands was divided by the average number of species in them all (Ewel 1984). A similarity value of 1.0 would be found if the wetlands had identical species assemblages.

Statistical analyses

Each year's results of peak biomass dry weights were compared in an analysis of variance (ANOVA) with each wetland as a different treatment. In addition, the peak biomass values for each wetland were compared for each year to determine whether productivity changed significantly from one growing season to the next. Because fewer quadrats were sampled in 1992, the comparison from one year to the next was based only on the data from the quadrats that were monitored every year.

Results

Successional trends

Floristic composition varied among the four experimental wetlands over the study period (Table 5). In the four wetlands, 112 different species were observed from 1988 through 1992. The types of species that became established ranged from facultative upland species (e.g., *Daucus carota*), to common obligate wetland species (e.g., *Typha latifolia*), to less common obligate wetland species (e.g., *Carex vulpinoidea*).

Three species were manually introduced into the wetland. Of these, only one became established. In early 1989, white water lily (*Nymphea odorata*) was planted in Wetlands 3 and 4; these populations expanded throughout the 1989 and 1990 growing seasons. American water lotus (*Nelumbo lutea*) was planted in Wetland 3 but did not emerge, probably because of dry soil conditions when the rootstocks were planted. A few individuals of water shield (*Brasenia schreberi*) were introduced into all wetlands except Wetland 5 during 1990, but did not survive beyond this growing season. All other species became established voluntarily.

The indicator status of each species as determined by the U.S. Fish and Wildlife Service (Reed 1988) is also shown in Table 5. Not surprisingly, changes in diversity were accompanied by a shift in the nature of the plants present. In 1988, the ratio of wetland (those with an obl or facw rating) to upland (facu or upl ratings) species was approximately 1:1, while nearly 100 percent of all species present after 1989 were wetland species (Figure 29). Conversely, the occurrence of obligate upland (upl) species decreased from an average of 24 percent in 1988 to 0 percent in 1990, 1 percent in 1991 and 8 percent in 1992.

Similarity values for vegetation, which indicate the proportion of species the four wetlands have in common, were calculated to be 53 percent in 1988, 34 percent in 1989, 52 percent in 1990, 59 percent in 1991, and 35 percent in 1992. The low value in 1989 is indicative of the reorganization occurring in the plant communities after flooding. Changing environmental conditions led to the extirpation of some species (e.g., *Oenothera biennis*) and establishment of others (*Leersia oryzoides*). The increasing similarity indices in 1990 and 1991 indicate that the structure of the wetlands' plant communities converged as the wetlands adapted to flooded conditions. The decrease in similarity value in 1992 may have been due to the change in hydrologic conditions, i.e., the fact that water flow into the wetlands stopped for 3 weeks from the end of June through the beginning of July, and flow varied over the summer to match the variation in river flow.

Peak biomass

There was a substantial range in total biomass production (as measured by peak biomass) in the four experimental wetlands (Figure 30). Macrophyte production increased each year in LFW 4 and 6. Growth in HFW 5 decreased in 1991 and then rose dramatically in 1992. This fall and subsequent rise was due to a shift in the community structure of HFW 5. In the first 2 sampling years, HFW 5 had large areas of *Phalaris arundinacea* that died back in 1991 and was not replaced until 1992, largely by *Typha* spp. Average peak biomass in Wetland 3 decreased after 1990. This decrease may have been because Wetland 3 had the deepest open areas (over 1 m); germination may have been inhibited because of light limitation.

For three of the four sampling seasons in which macrophyte peak biomass was measured, HFW 5 had the greatest average peak per square meter (1989, 1990, 1992); however, it was never significantly greater than peak biomass in LFW 4 ($p < 0.10$). In 1991, LFW 4 had a significantly higher peak biomass than any of the other wetlands ($p < 0.10$) because of its large *Typha* stand. LFW 6, the driest wetland because of water seepage to groundwater, had the lowest biomass until 1992, when the biomass in HFW 3 was lower (not significant at $p < 0.10$).

In 1991, after 2 years of flooding, *Typha* spp. and *Phalaris arundinacea* decreased in abundance and hence productivity under high flow conditions (HFW 3 and 5), so the initial surge in response to flooded conditions was reversed by continued deep water levels. At peak biomass, the average depth in HFW 3 was 69 cm, and in HFW 5, the mean depth was 60 cm. In LFW 4, the depth was slightly less at 57 cm, and mean depth in LFW 6 was 9 cm. In 1991, peak biomass in the high-flow wetlands fell considerably below 1990 levels. In HFW 3, far less cattail production was recorded in 1991; and in HFW 5, the dense cover of reed canary grass had virtually disappeared by the 1991 growing season. In 1992, biomass production in HFW 5 recovered, probably because the areas that had previously been vegetated by *Phalaris arundinacea* were replaced by areas of *Typha*.

Discussion

Species changes

In 1988 (dry conditions), a high of 29 species was recorded in Wetland 6, partly because of the growth of many upland species in this basin. Fifty-eight percent of the species in Wetland 6 in 1988 was ranked as either upland or facultative species (i.e., just as likely to be found in wet areas as dry areas; after Reed 1988). Because it was dry earlier in the growing season than the other wetlands, Wetland 6 was more prone to invasion by species that are often considered agricultural weeds such as velvet leaf (*Abutilon theophrasti*) and switch grass (*Panicum capillare*). The other wetlands had saturated soils for a longer period during the growing season and were not colonized to as great an extent by these species.

The highest diversity in a given site from 1989 through 1991 (wet conditions) was found in Wetland 4 in spite of the fact that nearly 50 percent of the wetland was dominated by cattail (*Typha* spp.). Many other species were interspersed with the cattail, and many species common to wet meadows were found along the edges of this wetland (e.g., *Carex* spp.). In 1992, Wetland 3 had the greatest diversity (58 species), even though it also had the lowest biomass.

Flooding the basins in 1989 represented a major disturbance to the plant communities, and it brought about rapid changes in their composition. Changes in species composition in response to flooding depend both on the type of vegetation originally present and the final depth of standing water. The various species that comprise a wetland community react differently to hydrologic change. A range of responses has been noted including declining species numbers, shifts in the patterns of carbon allocation, reduced productivity, and shifts in the mode of reproduction (Kadlec 1962; Lieffers and Shay 1981; Sjoberg and Danell 1983). Total species numbers decreased in the experimental wetlands immediately after flooding: thirty-five species were observed in 1988 and twenty-eight in 1989. As more wetland species invaded, the number of species increased to 35 in 1990, 36 in 1991, and 84 in 1992.

Successional trends that are a function of changes in the physical environment have been described in a qualitative model by van der Valk (1981). In this model, denoted "Gleasonian" to indicate the individualistic nature of species' response, the physical environment behaves as a sieve, allowing the persistence of species adapted to the conditions at hand. Changes in the plant communities at the Des Plaines site are aptly described by this model. The composition of the macrophyte community in the four experimental wetlands converged for the 3 years after flooding (1990 - 1992). The community structure is developing within the physical confines of the system using available plant sources (propagules available in the seed bank and those brought in by waterfowl, water, and wind).

Macrophyte biomass production: Flow conditions versus initial conditions

Macrophyte production showed no clear relationship to the experimental hydrologic inflows at the Des Plaines River site, perhaps because the dominant species (*Typha* spp.) was well established in several of the basins prior to pumping. Once established, *Typha* spp. persisted and spread vegetatively in all four wetlands. This life form makes a macrophyte community relatively resistant to change. In perennial species such as broad-leaved cattail (*T. latifolia*), most carbon allocation is to competitive structures or vegetative reproduction, making it a species that is replaced successionally over a very long time scale (Grace and Wetzel 1981). Similarly, studies of the response of wetland tree species to different hydrologic flows have indicated that many growing seasons may be necessary for differences in growth to appear (Straub 1984). The macrophyte communities at the Des Plaines River wetlands will require a period longer than 5 years to show differences in biomass production as a function of inflow rates.

Patterns of species replacement and biomass production seen at the Des Plaines River site indicate that early development of macrophyte communities in restored wetlands is a function of the initial composition of the plant communities and the "sieve" effect brought about by flooding. If wet-adapted, highly productive macrophytes are present at the time of flooding, they will persist, and productivity will rise as their coverage expands. If upland species are dominant, then flooding will decrease plant density and this biomass production. Wetland plants will colonize the areas formerly occupied by upland species; however, standing water often reduces the ability of aquatic plants to germinate, slowing the influx of new species. This pattern was evident at the site during the conversion period from dry to wet conditions (from 1989 to 1990). Wetlands that supported aquatic species such as cattail (*Typha* spp.) and reed canary grass (*Phalaris arundinacea*) prior to flooding had even larger areas dominated by these species after flooding. Spread of the highly productive species was reflected by higher biomass values in 1990 in all wetlands and increasing values in most of the wetlands after that.

Macrophyte biomass production: Water depth

Although LFW 4 and HFW 5 had similar water depths, biomass production did not decline in LFW 4 because it was dominated by cattail (*Typha* spp). *Typha* spp. are more able to persist in deeper water than reed canary grass (*Phalaris arundinacea*). It appears these species were responding to deepwater depth in the wetlands rather than to the flow differences. The presence of standing water, particularly at depths of up to 1 m as seen in HFW 3, may prevent germination, vegetative spread, and overall survival sufficiently to cause the 46 percent biomass reduction recorded there after 1990.

Ecosystem function of constructed versus natural wetlands

The relatively low biomass production in the experimental wetlands in 1990 and 1991 in comparison with natural wetlands (see Mitsch and Gosselink 1993) may be a function of the early developmental stage of the wetlands. Increased biomass in 1992 may indicate that the macrophyte growth in Wetlands 4, 5, and 6 has become denser and now more closely resembles macrophyte growth in a natural wetland. Successional changes in these wetlands seem to be driven more by water depth and antecedent conditions than by hydrologic flow conditions.

Conclusions

A clear divergence in peak biomass in the macrophyte communities of the four wetlands as a function of water inflow was not observed at the Des Plaines River site and may take longer than a few growing seasons to appear. Studies in forested wetlands indicate that the response time of long-lived species may be many years in length. This points to the existence of a continuum of response times based on life history traits or the life span of the species present. Concurrent studies of algal growth and water column productivity at the Des Plaines site (Chapters 4 and 5) support the hypothesis that the productivity of high-flow and low-flow wetlands will eventually diverge. In comparison with the macrophyte community, turnover time is rapid in the microbial community, leading to relatively rapid response times. Response of the macrophyte community is slow in comparison, yet quick relative to trees in forested wetlands.

Table 5
List of Species Found in Experimental Wetlands 3 - 6, 1988 - 1992

IS	Species	HFW 3					LFW 4					HFW 5					LFW 6				
		88	89	90	91	92	88	89	90	91	92	88	89	90	91	92	88	89	90	91	92
facu	*Abutilon theophrasti*			X			X					X					X				
facw	*Acer negundo*				X					X					X						X
facw	*Acer saccharinum*				X					X					X						X
facw	*Agrostis stolonifera var. palustris*									X					X						
obl	*Alisma plantago-aquatica*	X	X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
facu	*Ambrosia artemisiifolia*						X					X					X				
fac	*Ambrosia trifida*											X					X				
upl	*Aristidia oligantha*																				
facu	*Apocynum cannabinum*				X					X											
obl	*Aster firmus*																				X
nl	*Aster* spp.				X																
obl	*Asclepias incarnata*				X					X					X			X			X
obl	*Bidens cernua*																				X
facw	*Bidens frondosa*																				X
obl	*Bidens tripartita*																				X
obl	*Brasenia schreberi*		X					X											X		
obl	*Carex bebbii*				X			X													
facw	*Carex frankii*				X					X					X						
facu	*Carex normalis*				X					X					X						
obl	*Carex tribuloides*									X											
obl	*Carex vulpinoidea*				X			X	X	X	X				X						X
obl	*Ceratophyllum demersum*			X	X										X					X	X
obl	*Chara sp.*	X	X	X	X			X	X	X	X			X	X	X			X	X	X
fac	*Chenopodium album*											X			X		X				X
obl	*Cicuta maculata*				X																
obl	*Cladophora sp.*		X	X	X				X	X	X			X	X	X			X	X	X
upl	*Convolvulus arvensis*				X					X					X			X			X
facw	*Cornus stolonifera*				X					X					X						X
nl	*Cuscuta spp*																X				
obl	*Cyperus erythrorhizos*		X						X	X	X			X	X	X			X	X	
facw	*Cyperus esculentus*						X														
facw	*Cyperus ferruginescens*														X					X	
facw	*Cyperus odoratus*																				X
facw	*Cyperus strigosis*									X										X	
upl	*Daucus carota*						X					X					X				
facu	*Duchesnea indica*							X			X							X			
facw	*Echinochloa crusgalli*											X					X		X	X	
obl	*Eleocharis calva*		X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
obl	*Epilobium coloratum*				X											X					
facw	*Equisetum hyemale*				X																X
fac	*Erigeron annuus (daisy fleabane)*											X					X				
facw	*Eupatorium maculatum*				X																
facw	*Geum canadensis*				X					X											
facw	*Helenium autumnale*																				X
facw	*Impatiens capensis*				X																
obl	*Juncus acuminatus*								X												
obl	*Juncus canadensis*									X											
facw	*Juncus dudleyi*				X					X	X				X						
obl	*Juncus effusus*						X														
obl	*Juncus nodosus*				X					X					X						
obl	*Leersia oryzoides*		X		X					X	X				X						X
obl	*Lemna minor*		X	X	X				X	X	X			X	X	X			X	X	X
obl	*Ludwigia palustris*	X	X	X	X				X	X	X								X		
obl	*Lycopus americanus*				X				X	X	X			X	X	X	X			X	X

(continued)

Note: Each species' rating according to Reed (1988) is shown to indicate its wetland indicator status (IS), where obl = obligate wetland species; facw = facultative wetland species; fac = facultative species; facu = facultative upland species; upl = upland species; and nl = not listed.

Table 5 (Concluded)

IS	Species	HFW 3					LFW 4					HFW 5					LFW 6				
		88	89	90	91	92	88	89	90	91	92	88	89	90	91	92	88	89	90	91	92
facw	*Lythrum alatum*					X					X										
facw	*Lythrum salicaria*					X					X										
facu	*Matricaria matricaroides (pineapp)*											X									
fac	*Medicago lupulina*						X					X					X				
facu	*Melilotus alba (white sweet clover)*											X					X				
facw	*Mentha spp.*					X				X	X					X		X			X
facw	*Mentha arvensis var. villosa*																				X
facw	*Mentha piperita*										X										X
obl	*Mimmulus ringens*					X					X					X	X				X
obl	*Myriophyllum spicatum*		X	X	X			X	X	X					X	X				X	X
obl	*Nymphaea odorata*	X	X	X	X			X	X	X	X									X	X
facu	*Oenothera biennis*					X							X					X			
fac	*Panicum capillare*					X											X	X			
fac	*Panicum virgatum*					X					X					X					X
facw	*Phalaris arundinacea*	X	X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
obl	*Phyla lanceolata*															X					X
obl	*Polygonum amphibium*																				X
upl	*Polygonum aviculare*						X					X					X				
facw	*Polygonum lapathifolium*		X	X	X			X	X	X	X				X	X	X	X	X	X	X
obl	*Polygonum natans*		X	X	X				X	X	X			X	X	X			X	X	X
facw	*Polygonum pennsylvanicum*		X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
facw	*Polygonum persicaria*						X	X				X					X		X		
obl	*Pontederia cordata*													X							
fac	*Populus deltiodes*		X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
facu	*Populus tremuloides*				X																
obl	*Potamogeton crispus*		X	X	X				X												
obl	*Potamogeton foliosis*		X	X	X				X	X	X				X	X			X		
obl	*Potamogeton pectinatus*			X					X	X					X				X	X	
obl	*Ranunculus longirostris*				X					X	X										
obl	*Ranunculus scleratus*										X										
obl	*Rorippa palustris*						X	X				X	X					X			
nl	*Rubus sp.*										X										
fac	*Rumex crispus*					X	X						X				X	X			X
obl	*Sagittaria sp.*																		X		
obl	*Sagittaria latifolia*			X	X			X	X	X	X		X	X		X			X	X	X
obl	*Salix spp.*				X		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
facw	*Sambucus canadensis*										X										
obl	*Scirpus acutus*		X	X	X		X	X	X	X	X		X	X	X	X					X
obl	*Scirpus americanus*						X					X					X		X		
obl	*Scirpus atrovirens*		X		X					X	X					X					
obl	*Scirpus fluviatilis*	X	X	X																	
obl	*Scirpus lineatus*									X											
obl	*Scirpus validus*		X	X	X		X	X	X	X	X	X	X	X	X	X			X	X	X
facu	*Setaria glauca*																				X
obl	*Setaria viridis (foxtail)*																X				
fac	*Solanum dulcamara*																X				
facu	*Solidago canadensis*					X					X						X				
fac	*Sonchus arvensis*					X															
facu	*Sorghastrum nutans*															X					
obl	*Sparganium eurycarpum*																		X		
nl	*Spirogyra*		X	X	X				X	X	X			X	X	X			X	X	X
facw	*Stachys aspera*																				X
facw	*Teucrium occidentale*					X															
obl	*Typha angustifolia*	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
obl	*Typha latifolia*	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
obl	*Utricularia vulgaris*										X										
obl	*Vallisneria americana*		X	X				X	X					X		X			X		X
facu	*Verbena urticifolia*										X										
nl	*Xanthium pensylvanicum*																				X
	Total Number	2	9	26	25	58	21	19	28	34	53	22	14	20	22	43	29	17	26	27	48
	Total wetland species (obl + facw)	2	9	24	22	45	14	16	26	32	43	13	11	18	20	36	12	11	24	25	40
	Total facultative species (fac)	0	0	1	1	5	3	1	1	1	2	4	2	1	1	3	8	3	1	1	4
	Total upland species (facu + upl)	0	0	0	1	6	4	2	0	0	6	5	1	0	0	3	8	3	0	0	2
	No listing	0	0	1	1	2	0	0	1	1	2	0	0	1	1	1	1	0	1	1	2

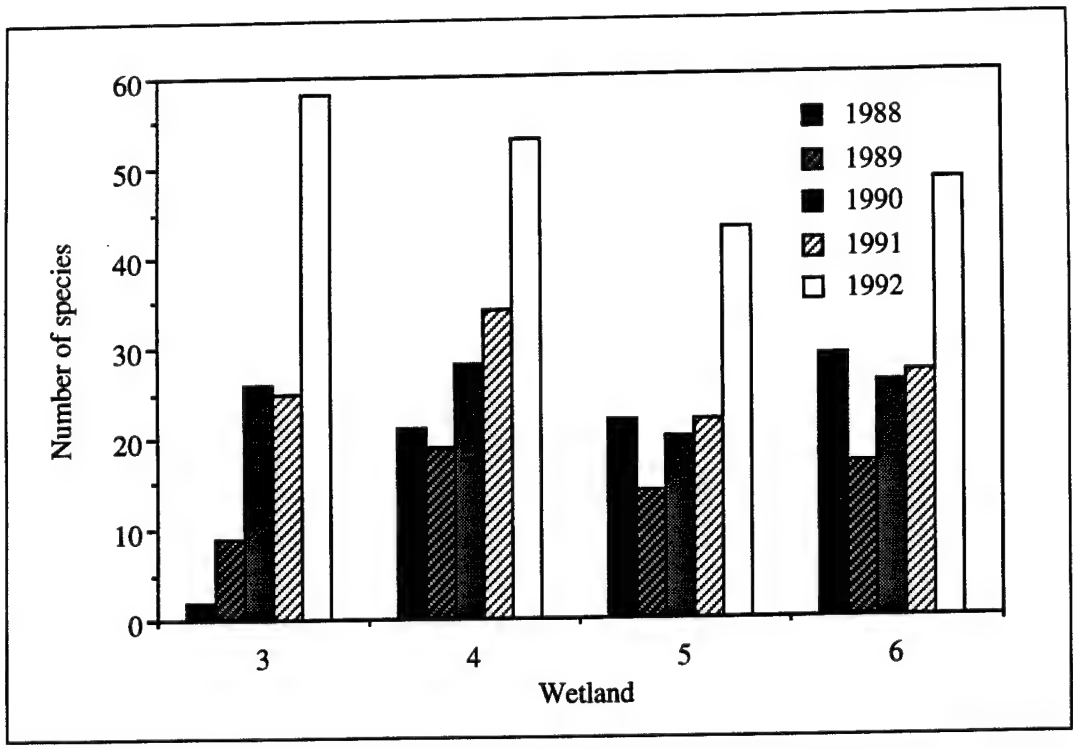

Figure 29. Proportion of species in wetlands that are designated wetland indicator species (those that are obligate wetland plants or facultative wetland plants) as defined by Reed (1988). Most vegetation was removed from Experimental Wetlands in 1988

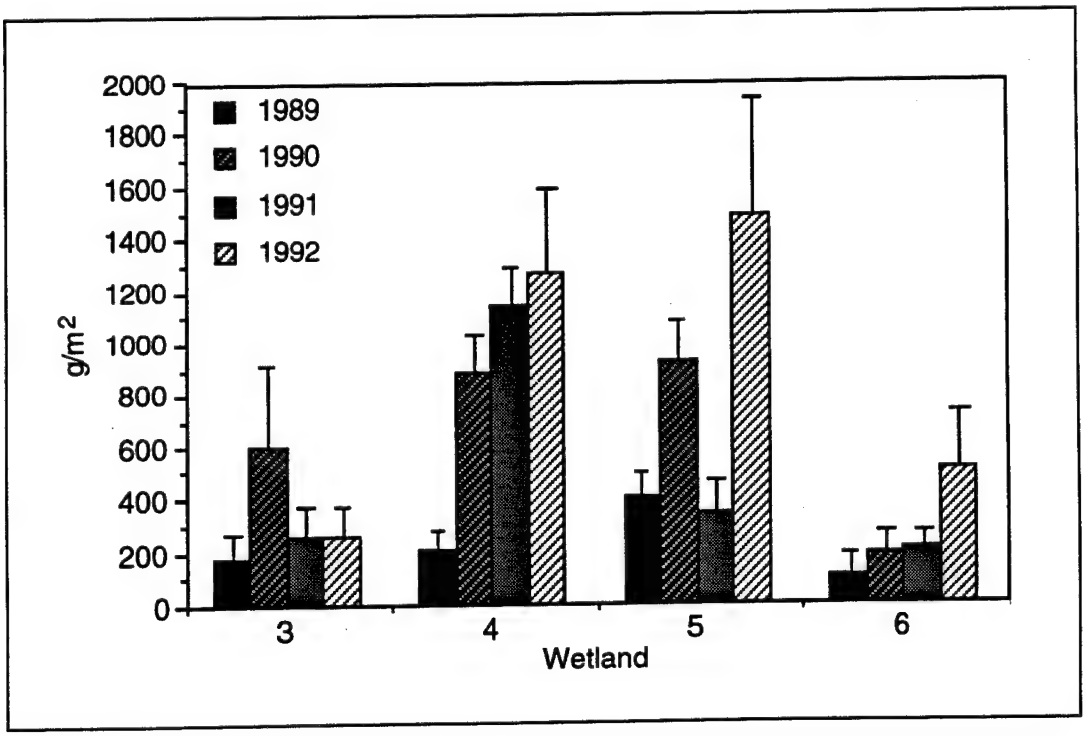

Figure 30. Peak biomass (average g dry weight/m^2) in experimental wetlands for the growing seasons of 1989 -1992

4 Periphyton Productivity on Artificial and Natural Surfaces in Four Constructed Freshwater Wetlands Under Different Hydrologic Regimes

Introduction

Algal growth on the surfaces of submerged macrophytes, rocks, and sediments provides a significant contribution to the total primary productivity of aquatic systems. The periphyton community's contribution to productivity can vary widely. In a summary of 11 lake studies, Wetzel (1983a) showed that the periphyton communities contributed from as little as 1 percent to as much as 60 percent of total lake productivity. In oceans, deep lakes, and downstream areas of rivers, phytoplankton dominates productivity, but when the sediment to water ratio increases, macrophytes and periphyton become more significant contributors to the system's productivity (Sand-Jensen and Borum 1991). Because of the usually shallow depth in wetlands and the many potential surfaces for colonization, the contribution of attached algae to wetland productivity is expected to be significant.

Many periphyton studies have been conducted in the littoral zones of lakes (Allen 1971; Cuker 1983; Millie and Lowe 1983; Cattaneo 1987; Burkholder and Wetzel 1989) and in streams (Elwood and Nelson 1972; Perkins and Kaplan 1978; Bott and Ritter 1981; Horner and Welch 1981; Hill and Webster 1982; Ennis and Albright 1982), but few studies of periphyton productivity have been conducted in freshwater wetlands (Brock 1970; Hooper and Robinson 1976; Haines, Rogers, and Rogers 1987). Interest in wetland retention of nutrients has inspired recent wetland periphyton nutrient uptake studies (Vymazal 1989; Vymazal and Richardson 1992). Attached algae are known to be significant contributors to the primary productivity of salt marshes, with contributions from one-fourth to one-third of macrophyte productivity in east coast salt marshes

(Pomeroy 1959; Gallagher and Daiber 1974; Van Raalte, Valiela, and Teal 1976) and 80 to 140 percent of macrophyte productivity in southern California salt marshes (Zedler 1980). The higher ratios recorded in California were due to both higher algal production and lower macrophyte production than usually seen in Eastern United States marshes. Twilley (1988) found epiphytic algae to contribute as much as 16 percent of the total carbon fixed in mangrove wetlands of Florida and Puerto Rico. The effect of water velocity on periphyton community structure and productivity has been the subject of many studies, most associated with simulations of stream ecosystems (Horner and Welch 1981; Traaen and Lindstrom 1983; Biggs and Close 1989; Horner et al. 1990). However, the influence of hydrologic flow (amount of water throughflow) on freshwater wetland periphyton productivity has remained unexplored.

To investigate and quantify the effect of two different hydrologic regimes (flow rates) on the periphyton productivity of constructed freshwater wetlands, a study of periphyton accumulation was conducted on both artificial surfaces and on macrophytes at the Des Plaines River Wetlands Demonstration Project in Lake County, IL. Samples were collected during the 1991 growing season in four wetlands subjected to two different hydrologic regimes to test the hypothesis that the wetlands with higher flow conditions would have greater periphyton productivity. It was hypothesized that one of the first indications of divergence among the wetlands according to different hydrologic flows would be in the algal community because of its rapid turnover time. To estimate the relative contribution of periphyton to the wetlands' water column primary productivity, these data were combined with those from a concurrent study of water column productivity (see Chapter 5).

Methods

Artificial substrata

Sampling. Artificial substrata (Figure 31) were placed at the inflow and outflow of the four wetlands, and periphyton accumulation was measured every 2 weeks throughout the 1991 growing season. The samplers were placed about 30 m from the inflow and 10 m from the outflow of each wetland on May 3, 1991. The distances from the inflow and outflow structures varied slightly so the samplers could be placed in open water areas to avoid complications from macrophyte shading. The slides were attached vertically to prevent solids from accumulating, a concern when using horizontal substrata (Aloi 1990). To eliminate depth differences and, as much as possible, light variation from the experiment (Jones and Mayer 1983), the upper crossbar of the frame was situated 10 cm below the water surface and maintained at this depth throughout the growing season. Twenty-four slides were attached to each frame, and three slides were randomly selected and removed on eight sampling dates from May through August. Each slide was placed in 250 mL of distilled water. The bottles were shaken to detach the loosely attached periphyton, and then the slides were removed and scraped with razor blades. Any additional periphyton removed in this

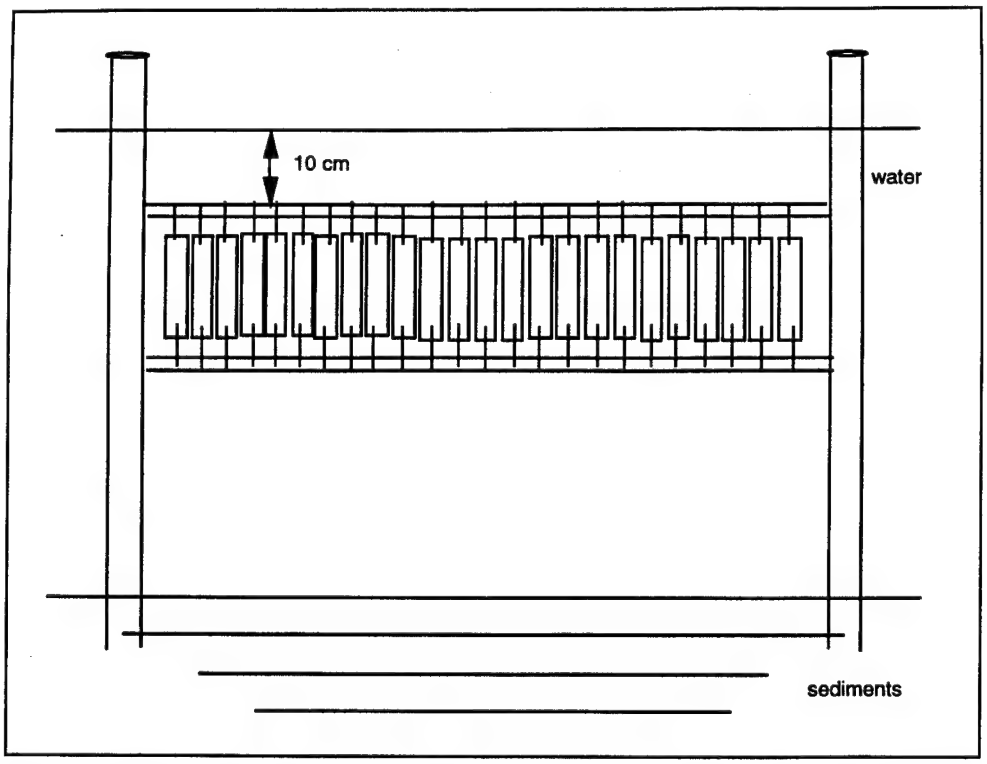

Figure 31. Periphyton sampler with 24 plastic microscope slides (36 cm^2) attached with nylon fishing line to a PVC frame and stand

manner were added to the bottle. Known volumes of the water/periphyton suspension were filtered, and filters were frozen until the time of analysis. Two filters (MSI cellulose 0.45-µm pore size) for each sample were kept for chlorophyll analyses, and two filters (Whatman glass microfiber, 0.45-µm pore size) per sample were used for dry weight and organic dry weight determinations.

Analysis. The material collected on the filters was analyzed for chlorophyll *a*, dry weight, and organic dry weight. The filters used for chlorophyll analyses were placed in centrifuge tubes with 10 mL of 90 percent acetone and stored at 4 °C for 24 hr. The tubes were centrifuged and the supernatant was analyzed for absorbance at 750, 665, 630, and 645 nm, using a 10-cm pathway and a Bausch and Lomb Model 600 spectrophotometer. Concentrations were calculated using Lorenzen's trichromatic equation given in Parsons, Maita, and Lalli (1984). Dry weight was estimated by drying preweighed filters at 100 °C for 24 hr. Filters were then weighed and combusted at 550 °C for 15 min. Weight lost in combustion was assumed to be the organic weight of the material present on the filter. All analyses were performed in duplicate.

Species composition of the periphyton community was determined once during the growing season. An extra slide was added to each sampler August 9 and collected on August 21. The added slides were stored in 95 percent alcohol and then examined under a microscope to identify the periphyton to genus.

Net biomass accumulation. The biomass accumulation rate (BAR) for each sampling interval was calculated as the change in organic dry weight divided by the time interval in days Δt:

$$BAR = \Delta ODW / \Delta t$$

where ODW is the organic dry weight (calculated with the average ratio of chlorophyll *a* to organic dry weight). The total net amount of biomass that accumulated during the growing season is estimated as the sum of the accumulation in all the sampling periods. It was calculated by plotting BAR at each sampler versus time and measuring the area under the curve. This amount was multiplied by 0.45 to give an estimate of carbon accumulation (Wetzel 1983a).

Data analysis. The mean values obtained for dry weight, organic dry weight, and chlorophyll *a* were compared using one-way ANOVA, with each wetland as a treatment, to determine whether there were differences among the wetlands. In a second set of analyses, in which differences within each wetland were examined, values from each inflow sampler were compared with values from the outflow of the same wetland in t-tests, using position within the wetlands as a treatment.

Natural substrata

Sampling and analysis. In addition to measurements of growth on artificial substrata, periphyton growth on underwater segments of the three dominant macrophyte species (*Typha* spp. *Polygonum* spp., and *Phalaris arundinacea*) was measured three times during the 1991 growing season (May 16, July 1, and August 16). Each sample consisted of three 15-cm shoot (*Polygonum* spp., and *Phalaris arundinacea*) or leaf sections (*Typha* spp.) that were harvested 10 cm below the water surface. Each sample was placed in a plastic bottle of 250 mL of distilled water. The samples included loosely attached algae so that productivity would not be underestimated (Haines, Rogers, and Rogers 1987). Three samples of three plant species in all four wetlands were gathered for a total of 36 samples per sampling date. Plants were chosen randomly throughout the wetlands so that position within the wetland was not a factor in this portion of the study.

The bottles were shaken to detach loosely attached algae, and the remaining epiphytes were removed from the macrophyte with a rubber spatula and added to those already in suspension. Known volumes of the suspension were filtered, and the filters were analyzed according to the same methods used for the artificial substrata. Once cleaned, the macrophyte segments were measured to determine the surface area available for epiphyte colonization.

Net biomass accumulation. Data collected from the macrophyte measurements (see Chapter 3) were used to provide an estimate of epiphyte net primary productivity. The total net biomass accumulation and carbon content were calculated in the same manner for epiphytes as for the periphyton growing on

artificial surfaces.

Peak epiphyte biomass. As part of a concurrent study on macrophyte productivity, water depth and the number of stems per square meter of *Typha* spp., *Polygonum* spp., and *Phalaris arundinacea* were measured at 16 permanent reference quadrats in May, July, and August (see Chapter 3). The average stem (*Polygonum* spp. and *Phalaris arundinacea*) or whole plant (*Typha* spp.) circumference was multiplied by the water depth to estimate the total surface area available in each quadrat (in $cm^2 \cdot m^{-2}$ of wetland). The surface area for each macrophyte species and quadrat was multiplied by the average organic dry weight ($mg \cdot cm^{-2}$) of epiphytes (estimated using chlorophyll *a*) growing on that species in that wetland. This gave an estimate of epiphyte biomass for each quadrat. The average of the quadrat totals represents the estimated peak epiphyte biomass production per square meter of each wetland ($mg \cdot m^{-2}$).

Data analysis. To determine whether epiphytic growth varied among the wetlands, the wetland means for each of the measured parameters were compared in one-way ANOVA with each wetland used as a treatment. To determine whether there were significant differences among the epiphyte communities growing on the three macrophyte species, chlorophyll *a*, dry weight, and organic dry weight were compared for each of the macrophyte species. For this comparison, the data from all the wetlands were combined and then regrouped according to macrophyte species, and the means were compared in one-way ANOVA, with macrophyte species as the treatment. This set of analyses was performed to determine whether there were differences among the macrophyte species in capacity to serve as an epiphyte substrate.

Results and Discussion

Artificial substrata

Seasonal trends. Periphyton chlorophyll *a* showed a peak early in the growing season (June 17) and again between August 9 and 21 with the exception of the results from the outflow of LFW 4, where levels remained low throughout the summer (Figure 32). Organic dry weight reached a maximum within the first three sampling dates (by June 17) at six of the eight sites. At the outflow of LFW 4 and the inflow of HFW 5, the maximum was reached by the fifth sampling date (July 15; Figure 33).

Dry weight peaked later than the other two parameters and then decreased throughout the rest of the growing season (Figure 34). The later peak in dry weight may have been due to an increase in the solids retention capacity of the periphyton's surface. As the algae grew and then senesced, more binding sites for waterborne clay particles were created and more inorganic material could accumulate on the samplers. The slightly downward slope of the curves toward

the end of the season may indicate a period of sloughing of dead organic material and solids. Dry weight on the slides showed a significant linear relationship with total suspended solids in the water column; however, the variability was high ($r^2 = 0.25$, $n = 180$, $p = 0.10$).

In all four wetlands, the peak accumulation of each parameter was of the same magnitude and occurred within 1 month. This rate lies within the time range observed in many studies where colonization of introduced substrata occurred at an exponential rate for the first 2 weeks of exposure and then slowed (Kevern, Wilhm, and Van Dyne 1966; Lamberti and Resh 1985; Paul and Duthie 1988; Aloi 1990). The seasonal trends for periphyton growth observed here corresponded to the bimodal pattern seen in a number of other periphyton studies (Van Raalte, Valiela, and Teal 1976; Sand-Jensen and Søndergaard 1981; Roos 1983; Perrin, Bothwell, and Slaney 1987). After the first peak, periphyton growth is often curtailed because of self-shading and lack of space (Sand-Jensen and Borum 1991). The second peak in chlorophyll *a* observed in August may have been due to changing nutrient availability and a resulting shift in the algal community.

The occurrence of algal genera present on slides placed on the samplers in August are listed in Table 6. The most commonly seen genus, *Cladophora*, was present throughout the water column of all of the wetlands, as well as on the samplers. *Fragilaria* was the most common diatom genus noted. The relatively low diversity may be an indicator of the system's immaturity.

Comparisons among the wetlands. As a general pattern, the average results for all of the parameters followed the order: high flow 5 > high flow 3 = low flow 6 > low flow 4. Periphyton chlorophyll *a* results indicated a significant difference between the two hydrologic treatments (Figure 35a). The two HFW 3 and 5 showed significantly higher chlorophyll *a* densities than did the LFW 4 and 6 ($p < 0.10$). The two hydrologic flows did not bring about clear differences in organic dry weight accumulation since results were dissimilar within the high flow treatment (Figure 35b). The highest levels of organic dry weight were measured in HFW 5. Dry weight accumulation showed proportional results to organic dry weight. The samplers in HFW 5 had significantly greater average dry weight per surface area than the samplers in the other wetlands (Figure 35c).

The calculation of total net carbon accumulation showed that the samplers in HFW 5 accumulated the greatest amount of total organic carbon (32 mg $C \cdot cm^{-2}$ of slide area·growing season^{-1}), while the other three averages were virtually the same (18 mg $C \cdot cm^{-2}$·growing season^{-1}; Table 7). HFW 5 may have had the highest average because residence time within HFW 5 (the smallest wetland) was shortest; therefore, nutrients were still available at the outflow, and periphyton growth there was not nutrient limited.

Comparisons between inflow and outflow sampling stations. After the

Table 6
Occurrence of Algal Genera on Periphyton Samplers: Colonization Between August 9 and 21, 1991 (keys: APHA 1989; Prescott 1978)

	HFW 3		LFW 4		HFW 5		LFW 6	
	in	out	in	out	in	out	in	out
Bacillariophyceae								
Fragilaria	X	X			X			X
Stauroneis	X	X	X		X	X		X
Navicula	X							
Pinnularia		X	X		X	X	X	
Chlorophyta								
Cladophora	X	X	X	X	X			X
Ankistrodesmus	X		X		X	X	X	
Chlorella		X					X	X
Pediastrum			X	X	X			
Bulbochaete			X					
Spirogyra				X				
Cyanophyta								
Anabaena	X	X	X			X	X	

first two sampling dates, measurements for all the parameters tended to be greater at inflow sampling stations than at outflow stations (Figures 32 to 34). When the data for the entire growing season were averaged, measurements at the inflow were generally greater than at the outflow sites (Figure 36). Lower nutrient concentrations at the outflows may have placed a stress on periphyton productivity there. Weekly water samples taken at the inflow and outflow of each wetland indicated that nutrient concentrations decreased significantly before the water reached the outflow. On average, TP concentrations in 1991 decreased by 120 $\mu g\text{-}P \cdot L^{-1}$ between the inflow and outflow (80 percent decrease); SRP decreased by 20 $\mu g\text{-}P \cdot L^{-1}$ (see Chapter 2); nitrate-nitrogen and total nitrogen both decreased by 1 $mg \cdot L^{-1}$ (80 and 60 percent decrease, respectively). Therefore, periphyton may have been nutrient limited at the outflow of these wetlands. Biomass at the outflow of LFW 4 was far lower than at any other sampling site. Periphyton growth there may have been more nutrient limited than at other sites because of the lower nutrient supply, which is, in turn, because of the low flow regime.

In LFW 6, the results from the inflow and outflow samplers did not differ significantly for dry weight or organic dry weight. By mid-June, parts of LFW 6 were dry because of groundwater seepage, and water did not flow continuously from inflow to outflow. Both of the samplers were submerged, but because the water was so shallow, the bottom crossbar of the sampler touched the sediments. The proximity to the sediments may have led to higher readings, particularly for dry weight accumulation.

The average of total net carbon fixation for the growing season for all of the

inflow sites (24 mg C·cm^{-2} of slide area·growing season^{-1}) was greater than that for the outflow sites (18 mg C·cm^{-2}); however, this result is due mostly to the exceptionally low reading at the outflow of LFW 4 (Table 7).

Natural substrata

An analysis of epiphytes collected from macrophytes shows differences among the wetlands were slight and are not consistent with hydrologic treatment (Figure 37). However, significant differences in the amount of epiphyte growth on each species were observed (Figure 38). Dry weight and organic dry weight were highest on *Polygonum* spp. and lowest on *Typha* spp. Epiphyte chlorophyll *a* was greater on *Phalaris arundinacea* and *Polygonum* spp. than on *Typha* spp.

Less dry weight and organic dry weight per unit area accumulated on artificial substrata than on natural substrata. However, average chlorophyll *a* per unit area was slightly higher on the slides than on the macrophytes. The heterogeneous nature of the macrophyte surfaces probably made them more conducive to dry weight accumulation because more microsites for attachment are available per unit of surface area. Organic dry weight included not only periphyton, but consumers as well. The macrophytes may have supported a larger population of heterotrophs than the slides. Average chlorophyll *a* may have been slightly higher on the slides because periphyton there were less light limited than on the macrophytes, where they were shaded not only by the macrophytes, but by the greater amount of dry weight.

Epiphyte peak biomass. Epiphyte biomass accumulation peaked on different dates in each wetland, and the estimated peak of epiphyte growth showed a wide range (from 50 mg dry weight·m^{-2} of wetland area in LFW 6 to 6,000 mg·m^{-2} in HFW 5). The differences do not appear to be the result of hydrologic flow. Peak growth of periphyton in HFW 5 was greatest; however, periphyton growth

Table 7
Estimate of Total Amount of Organic Carbon Fixed by Periphyton on Samplers over Growing Season (mg organic carbon·cm^{-2} of slide area) and Averages for Each Wetland and for Inflow and Outflow Samplers

Wetland	Inflow	Outflow	Average for wetland
HFW 3	24	12	18
HFW 5	28	36	32
LFW 4	32	3	18
LFW 6	14	22	18
Average Inflow: 24		Average Outflow: 18	

in HFW 3 (1,900 $mg \cdot m^{-2}$) and LFW 4 (3,200 $mg \cdot m^{-2}$) did not follow this pattern. The peak epiphyte growth depended more on the surface area available and on water depth than on the amount of water throughflow. Surface area and depth varied widely among the wetlands; the estimated available surface area ranged from 600 $cm^2 \cdot m^{-2}$ of wetland in LFW 6 to nearly 20,000 $cm^2 \cdot m^{-2}$ of wetland in LFW 4, and the average water depth was between 25 cm (LFW 6) and 71 cm (HFW 3).

Since the estimate of peak epiphyte growth was calculated using the number of stems of macrophytes per square meter in each wetland, and LFW 4 exhibited by far the densest growth, it provided the greatest amount of surface area to periphyton of the four wetlands. However, the macrophytes there supported less epiphyte growth per unit area than the macrophytes in the other wetlands, perhaps because the thick *Typha* canopy limited epiphyte growth through shading. Peak epiphyte production in LFW 6 was quite low. Macrophyte growth there was only slightly less dense than in the two HFW (see Chapter 3), but much of LFW 6 was dry in 1991 because of water losses to groundwater and drought conditions. The lack of wet surfaces in LFW 6 explains the lack of epiphyte biomass.

Wetland periphyton productivity

Epiphyte growth on macrophytes was estimated to contribute from 1 percent to 65 percent of the total water column primary productivity of these wetlands during the growing season. The lowest level was in LFW 6 where epiphyte growth was limited by a lack of wet surface area, and the highest contribution was in HFW 5. In HFW 3, the percent contribution of epiphytes to water column productivity was 36 percent and in LFW 4, it was 44 percent (Cronk 1992). Periphyton productivity of these wetlands was comparable with that of other water bodies (Table 8), particularly with similar substrates (i.e., macrophytes).

The estimated values of periphyton productivity on the slides were also comparable with results of other studies using artificial substrata (Table 9). Lai (1977) added nutrients to Lake Michigan water, and the amount of periphyton growth was similar to that at Des Plaines. The range of values in Lai's study indicates a response to water velocity differences, with the lowest productivity in stagnant water and the highest productivity at a water velocity of 11 $cm \cdot sec^{-1}$.

Hydrologic treatment and periphyton accumulation. HFW periphyton samplers had significantly greater average amounts of chlorophyll *a* per unit area than LFW. This may be in response to higher nutrient loadings in these wetlands. The distinction between the HFW and LFW did not hold true in the natural substrata portion of the study.

The results for dry weight and organic dry weight per sampler surface area do not indicate a clear response to the different flow treatments. In general, HFW 5 showed the highest values, and LFW 4 generally had the lowest measurements.

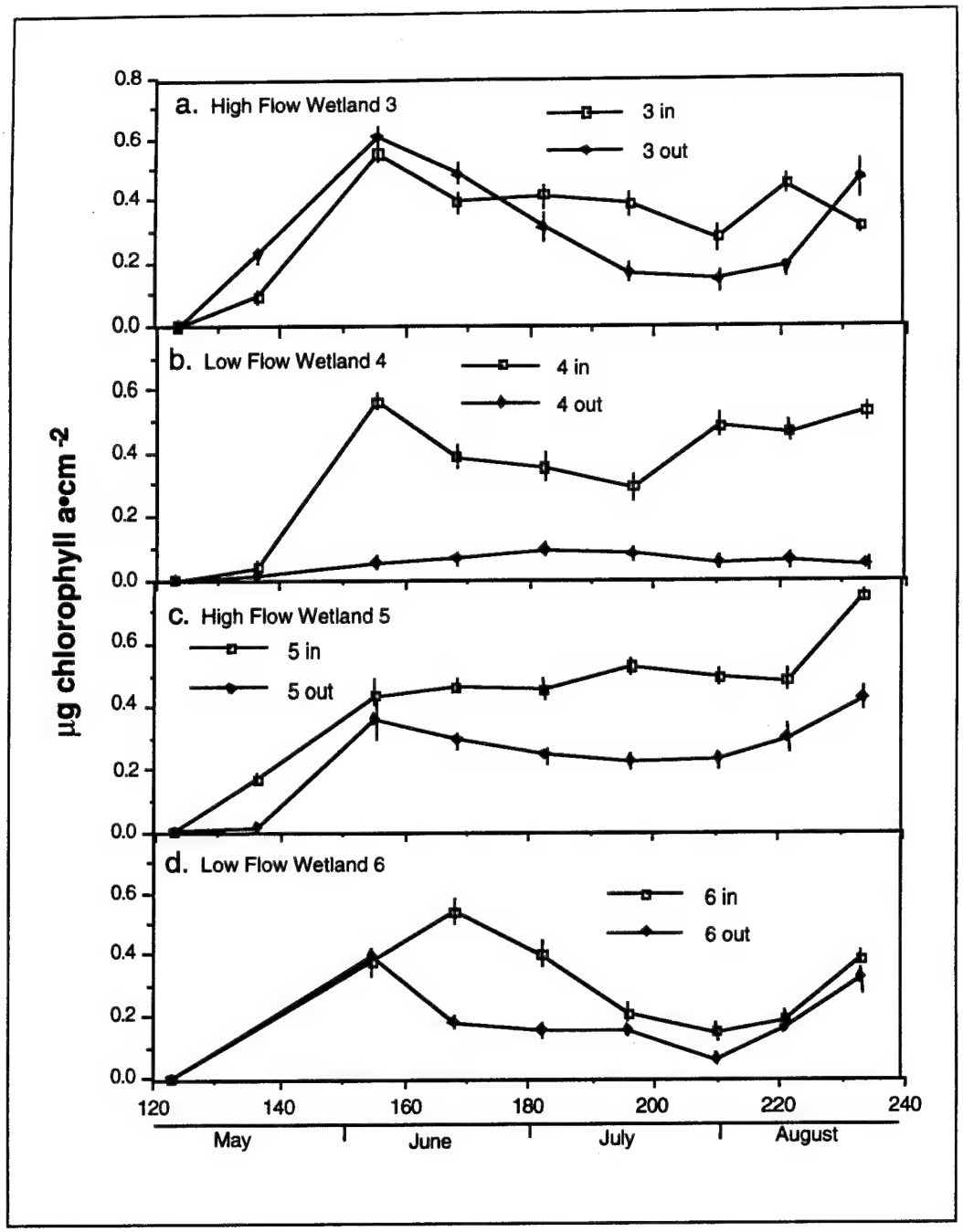

Figure 32. Average chlorophyll *a* in $\mu g \cdot cm^{-2}$ with standard error bars versus Julian date for periphyton collected from slides May 4 - August 21, 1991. Results are from samplers at inflows and outflows of (a) HFW 3; (b) LFW 4; (c) HFW 5; and (d) LFW 6

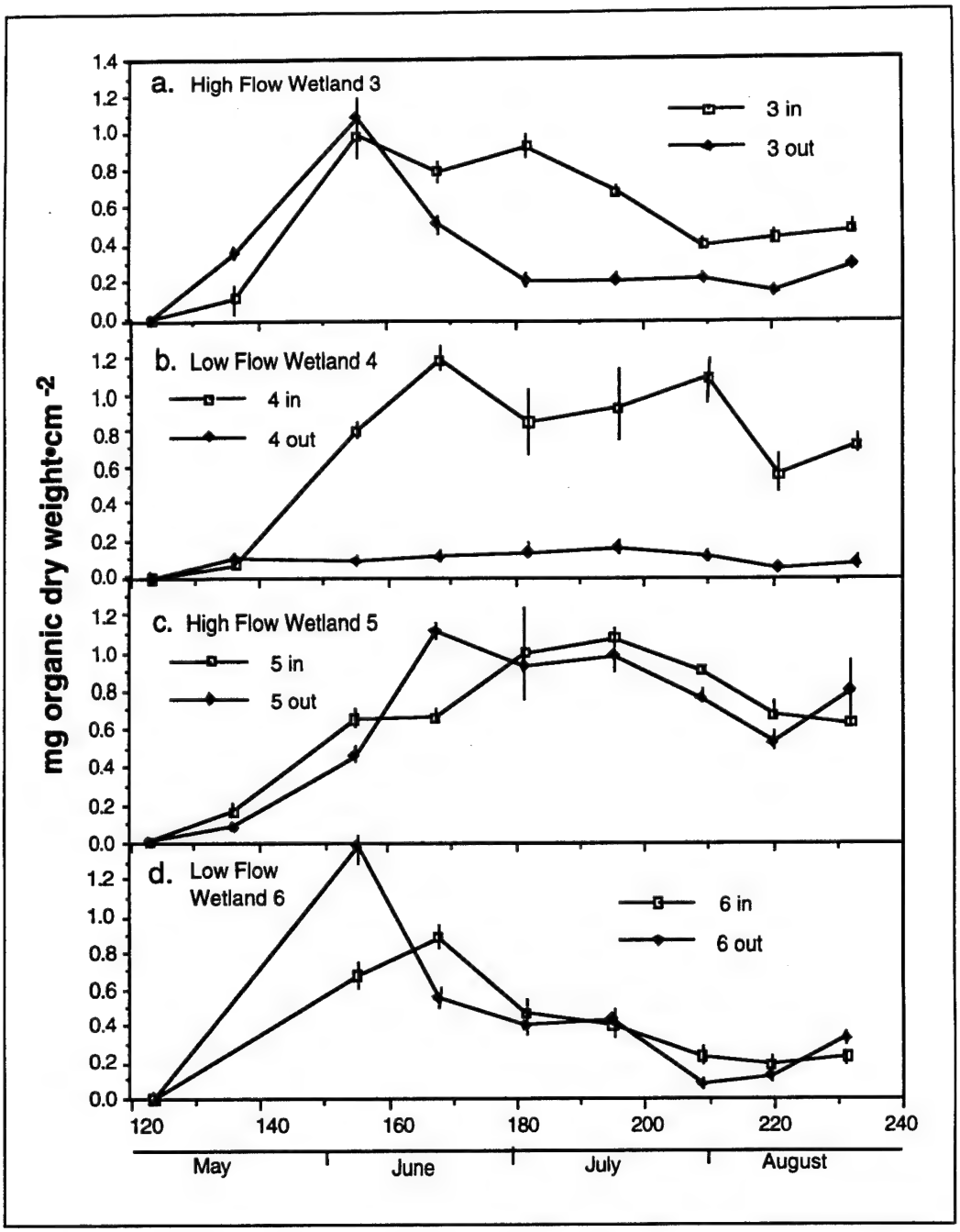

Figure 33. Average organic dry weight in $mg \cdot cm^{-2}$ with standard error bars versus Julian date for periphyton collected from slides May 4 - August 21, 1991

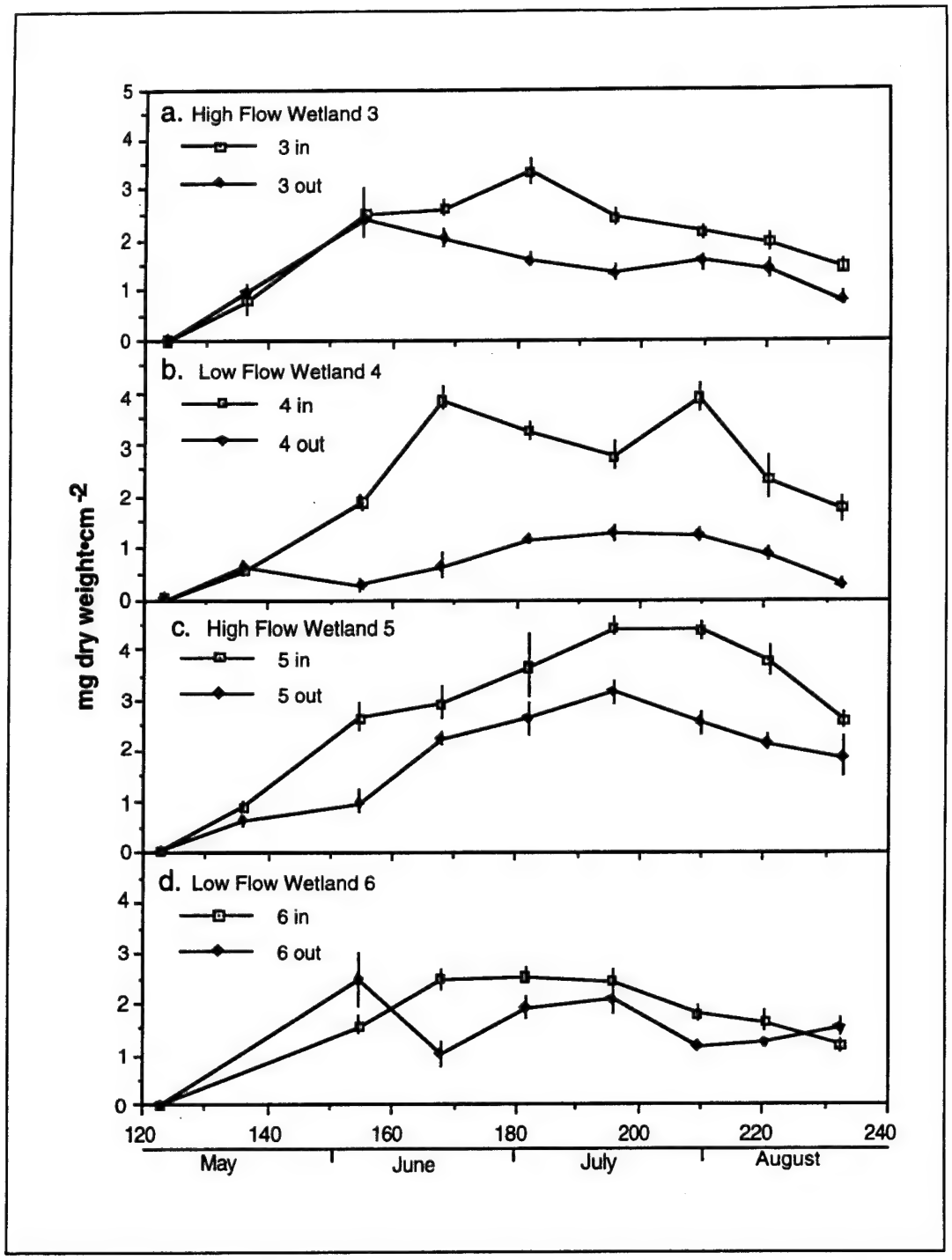

Figure 34. Dry weight in $mg \cdot cm^{-2}$ with standard error bars versus Julian date for periphyton collected from slides May 4 - August 21, 1991.

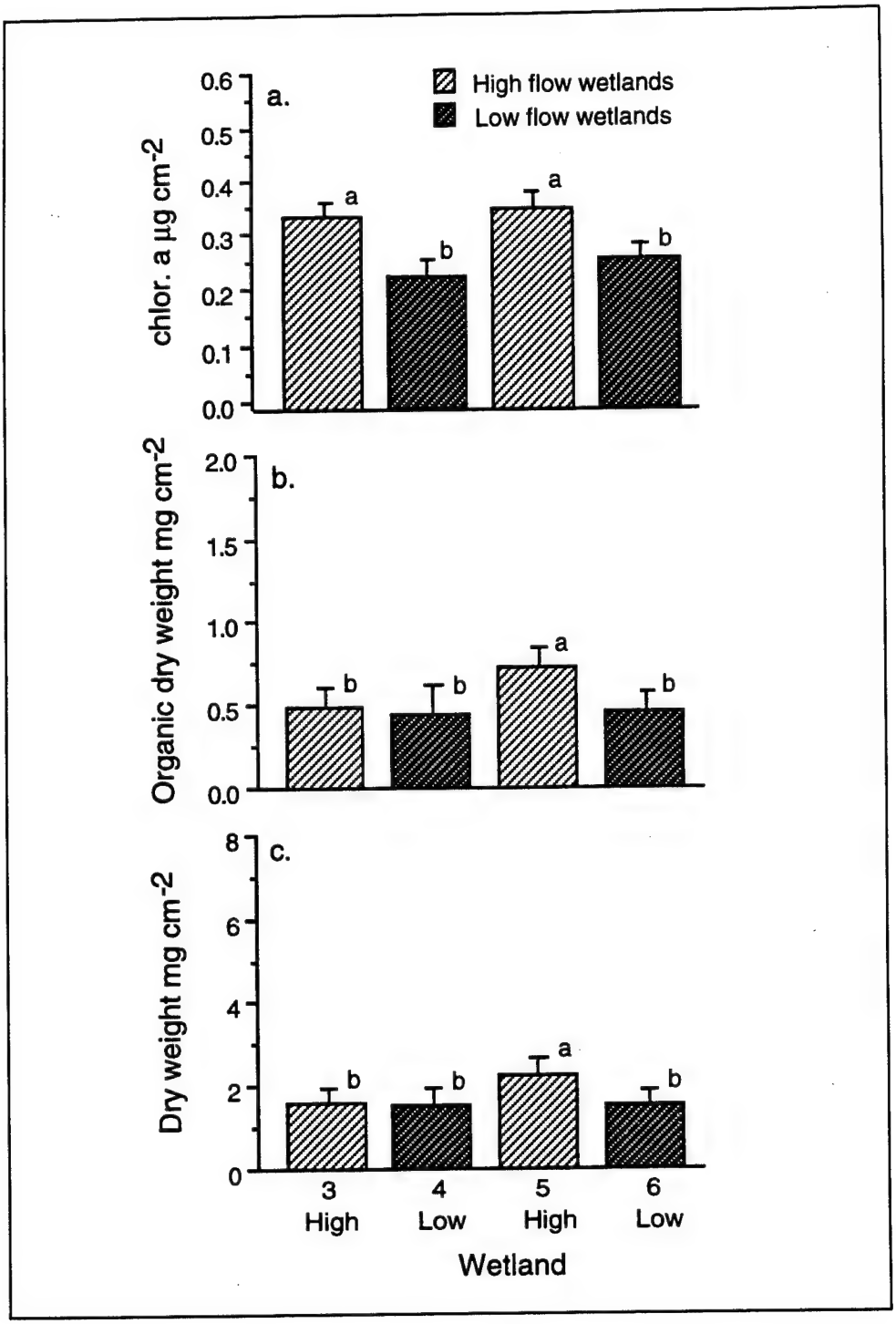

Figure 35. Average (a) chlorophyll *a* (µg·cm^{-2}), (b) organic dry weight (mg·cm^{-2}), and (c) dry weight (mg·cm^{-2}) of periphyton collected from slides in each wetland from May 16 - August 21, 1991. Bars indicate standard error and unlike letters indicate significant differences at $p < 0.10$. Sample size: (a) and (b) HFW 3 and LFW 4, n=46; HFW 5, n = 47; LFW 6, n = 42. (c) HFW 3, n = 46; LFW 4, n = 45; HFW 5, n = 47; LFW 6, n = 42

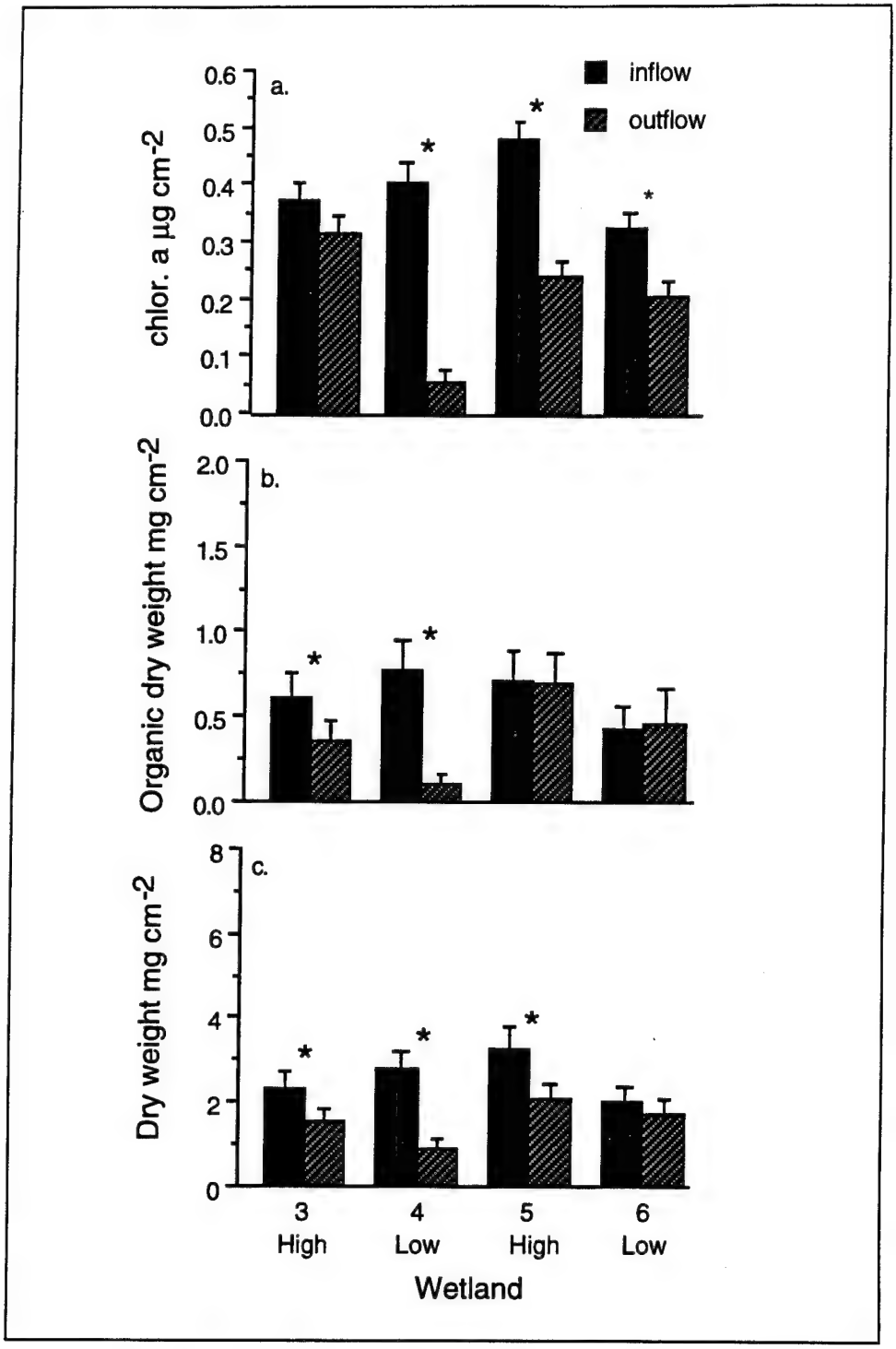

Figure 36. Average (a) chlorophyll *a* ($\mu g \cdot cm^{-2}$), (b) organic dry weight ($mg \cdot cm^{-2}$), and (c) dry weight ($mg \cdot cm^{-2}$) of periphyton collected from each sampler from May 16 - August 21, 1991. Bars indicate standard error and a * indicates a significant difference at $p < 0.10$. Sample size: n = 24 except: (a) HFW 3 in and LFW 4 in, n = 22; HFW 5 out, n = 23; LFW 6 in and out, n = 21. (b) HFW 3 in and out, and HFW 5 out, n = 23; LFW 6 in and out, n = 21. (c) HFW 3 in and LFW 4 out, n = 23; LFW 6 in, n = 21; LFW 6 out, n = 20

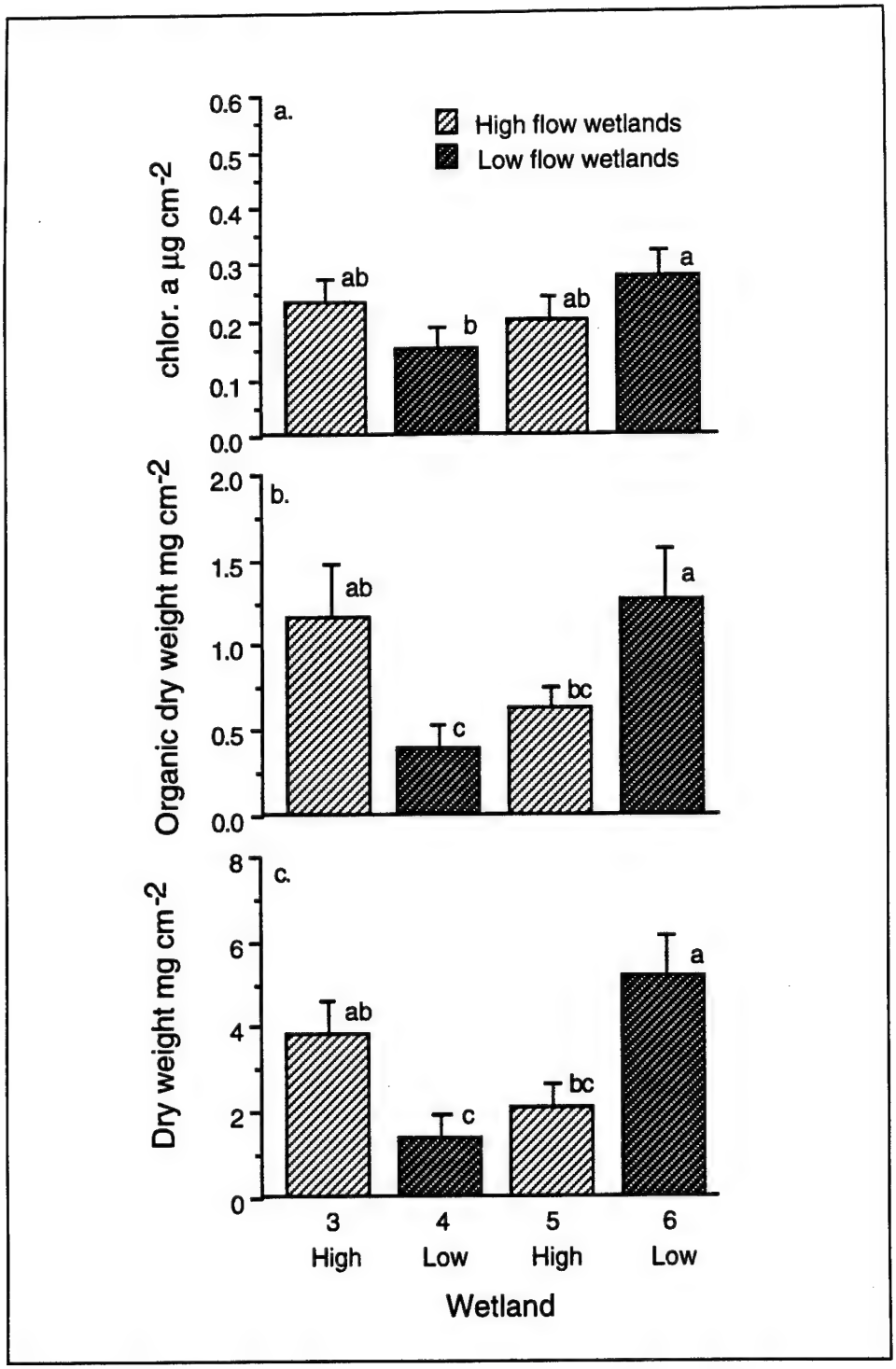

Figure 37. Average (a) chlorophyll *a* (µg·cm^{-2}), (b) organic dry weight (mg·cm^{-2}), and (c) dry weight (mg·cm^{-2}) of periphyton collected from macrophytes during the 1991 growing season. Bars indicate standard error and unlike letters indicate significant differences at $p < 0.10$; $n = 27$

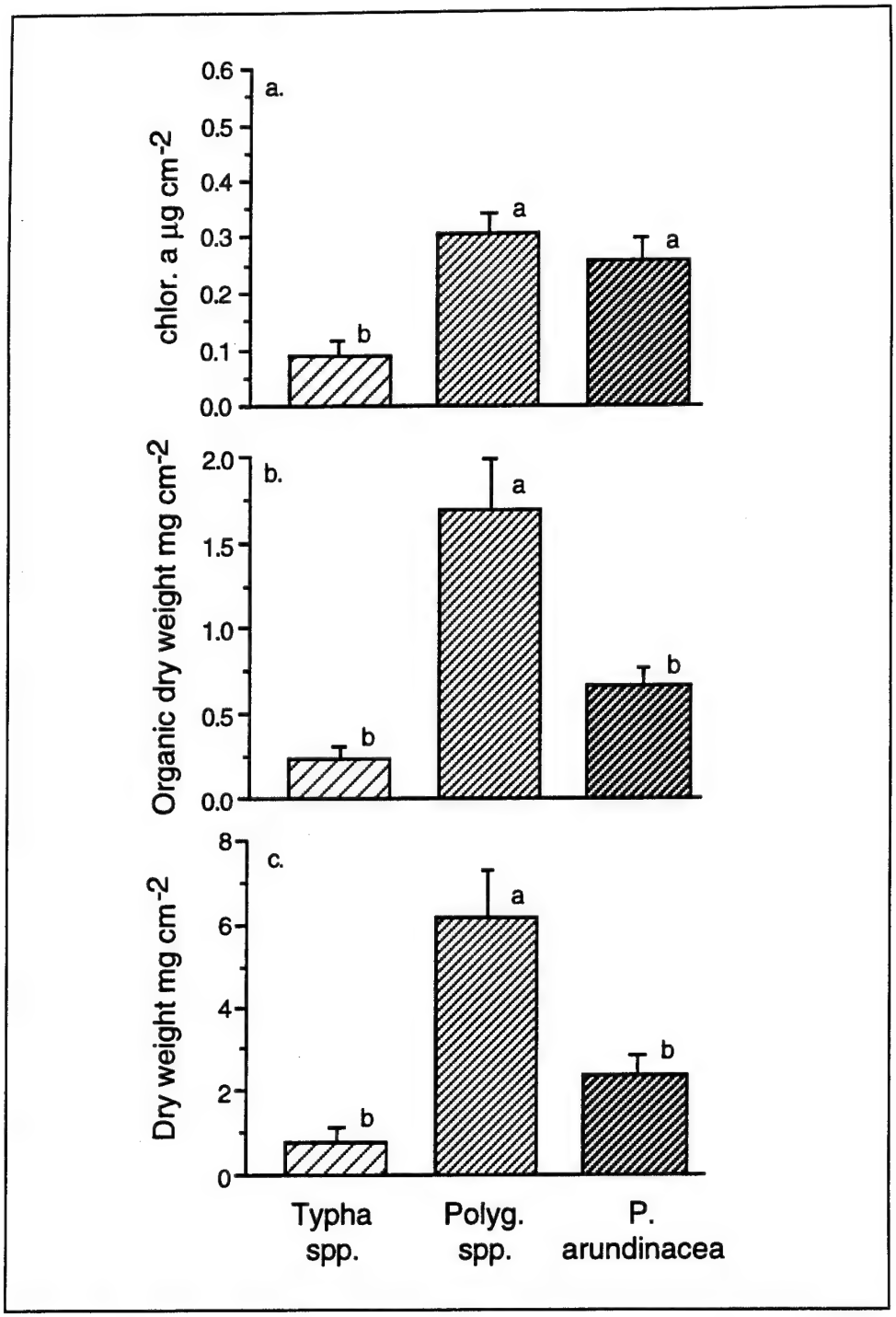

Figure 38. Average (a) chlorophyll *a* (µg·cm^{-2}), (b) organic dry weight (mg·cm^{-2}), and c) dry weight (mg·cm^{-2}) of periphyton collected from three different macrophyte types (*Typha* spp., *Polygonum* spp., and *Phalaris arundinacea*) during the 1991 growing season. Bars indicate standard error and unlike letters indicate significant differences at $p < 0.10$; $n = 36$

HFW 3 and LFW 6 had similar averages and were often more similar to one another than to the wetlands of similar hydrologic treatment.

In the macrophyte portion of the study, HFW 5 had the highest estimated peak epiphyte growth because it had the greatest amount of the macrophytes that bore more epiphytes (*Polygonum* spp. and *Phalaris arundinacea*). Epiphyte growth per unit area of macrophyte was greater in LFW 6 and HFW 3; but LFW 6 had fewer wet surfaces, and HFW 3 had fewer *Polygonum* spp. and *Phalaris arundinacea.*

The fact that average measurements in HFW 5 and LFW 4 were generally at the highest and lowest extremes suggests some response to the treatment. However, periphyton in HFW 3 and LFW 6 did not respond in the same manner. HFW 3 is deeper, larger, and has more areas of open water than HFW 5. LFW 6 is the largest wetland and had large areas that were dry during most of the growing season. Therefore, it is difficult to draw conclusions about the effect of the high or low flow treatment on periphyton dry weight and organic dry weight accumulation.

Epiphyte/macrophyte interactions. Macrophytes and epiphytes of freshwater systems may interact in at least two ways (Wetzel 1983b): chemically, through the diffusion of ions between the macrophytes and the epiphytes, i.e., one may provide nutrients for the other (Cattaneo and Kalff 1979; Carignan and Kalff 1982; Riber, Sorensen, and Kowalczewski 1983; Moeller, Burkholder, and Wetzel 1988); and physically, in that a macrophyte provides a substrate for epiphyte growth, but it also shades its own stem and makes light less available to the epiphytes as the macrophyte grows (Van Raalte, Valiela, and Teal 1976; Zedler 1980; Macauley, Clark, and Price 1988; Sand-Jensen, Borge, and Jeppesen 1989), or the macrophyte may inhibit epiphyte colonization by the presence of an exterior hydrophobic cuticle which degrades as the plant senesces (Goldsborough and Hickman 1991).

The difference in the amount of periphyton growing on each of the three macrophyte species in this study could be due in part to the growth habit of the macrophytes. *Polygonum* spp. extends horizontally through the water column and appeared to support more loosely attached periphyton than *Typha* spp. and *Phalaris arundinacea*. These present a more vertical substrate and may inhibit periphyton through self-shading.

Studies indicate that epiphytes may glean some nutrients from macrophyte surfaces, particularly if water nutrients are in short supply (Cattaneo and Kalff 1979; Carignan and Kalff 1982; Moeller, Burkholder, and Wetzel 1988). If the epiphyte community is able to acquire any of its nutrients from the underlying substratum, epiphyte growth may be favored on *Polygonum* spp. because of nutrient availability. In a 1990 study of the nutrient content of plant tissue at the Des Plaines River site, Fennessy (1991) found that *Polygonum* spp. had higher tissue nutrient concentrations than *Typha* spp. and *Phalaris arundinacea*. The mean nitrogen concentration of *Polygonum* spp. was 28 mg $N \cdot g^{-1}$ tissue, and 18

Table 8
Periphyton Productivity in Grams of Carbon per Unit Area per Year on Natural Substrates in Different Water Bodies

Substrate Type	Location	Periphyton Productivity g C·m^{-2}·year^{-1}	Source
Freshwater Wetlands			
Macrophytes (HFW 3)	Illinois	53	This study
(HFW 5)		85	
(LFW 4)		52	
(LFW 6)		2	
Salt Marshes			
Sediments	Georgia	148	Pomeroy 1959
	Delaware	80	Gallagher and Daiber 1974
	Massachusetts	47	Van Raalte, Valiela, and Teal 1976
	California	185-341	Zedler 1980
Lakes			
Macrophytes	Michigan	38	Wetzel et al. 1972
Macrophytes	New Hampshire	0.06	Jordan and Likens 1975
Phragmites communis	The Netherlands	67	Meulemans 1988

Table 9
Periphyton Productivity in Grams of Carbon per Unit of Substrate Area per Year on Artificial Substrates in Different Water Bodies

Substrate Type	Location	Periphyton Productivity (mg C·cm^{-2} of substrate·year^{-1})	Source
Freshwater Wetlands			
Plastic slides (HFW 3)	Illinois	18	This study
" (HFW 5)		32	
" (LFW 4)		18	
" (LFW 6)		18	
Lakes			
Metal, rocks	Illinois	2-19	Lai 1977 (microcosm using Lake Michigan water)[1]
Acrylic rods	Alberta, Canada	7-24	Goldsborough and Hickman 1991[2]

[1]Assumes g Cl=l0.45*g dry weight (Wetzel 1983a) and a 110-day growing season.
[2]Calculated using a C:chlorophyll a ratio of 40 (Sand-Jensen,Borg, and Jeppesen 1989).

mg $N \cdot g^{-1}$ for both *Typha* spp. and *P. arundinacea*. For phosphorus, the mean concentrations were 2.6 mg $P \cdot g^{-1}$ for *Polygonum* spp., 2.0 mg $P \cdot g^{-1}$ for *Typha* spp., and 1.5 mg $P \cdot g^{-1}$ for *P. arundinacea*. Whether the difference in substratum suitability was due to physical or chemical factors, the results indicate the necessity to examine several macrophyte species to determine the epiphytic contribution to the system's productivity.

Significance of periphyton in constructed freshwater wetlands

Periphyton contributed to nutrient uptake from the water column during the growing season. Biotic removal of nutrients from the water column is due, to a great extent, to the rapid turnover rate of phytoplankton and periphyton. Assuming a growing season from April 1 through October 1 and that the phosphorus content of epiphytes is 0.8 mg P·g organic dry weight of algae (Mitsch et al. 1992), then the average phosphorus uptake by epiphytes ranged from about 1 to 3 mg $P \cdot m^{-2}$ of wetland area·$week^{-1}$. This value is from 1 percent to 90 percent of the phosphorus uptake calculated for the water column phototrophic communities as a whole (see Chapter 2). The epiphytes are therefore significant users of phosphorus, and more periphyton would lead to the uptake of greater amounts of water column available nutrients.

In another study, periphyton growing on artificial substrata removed 80 percent of incoming ammonium, 70 percent of incoming SRP, and 15 percent of the nitrate from river water within 15 days of exposure (Vymazal 1989). The average nutrient levels were all considerably higher than at the Des Plaines River Wetlands. Given such high nutrient removal rates, periphyton might be used to treat nutrient enriched water. In general, periphyton grows more slowly than phytoplankton, but periphyton communities have nonetheless exhibited a greater relative response to nutrient inputs than phytoplankton in some studies (Van Raalte, Valiela, and Teal 1976; Philips, Eminson, and Moss 1978; Sand-Jensen and Søndergaard 1981; Peterson et al. 1985; Carrick and Lowe 1988; Fairchild and Everett 1988). The periphyton response may indicate that periphyton is more nutrient limited than the phytoplankton community.

Periphyton may also have acted as a filter for waterborne solids. Dry weight accumulation on the samplers at the inflows was greater than at the outflows (significantly greater in HFWs 3 and 5 and LFW 4 at $p < 0.10$), perhaps because inflow samplers were in areas of higher turbidity and total suspended solids concentrations than the outflow samplers (Cronk 1992).

Because of their capacity to remove nutrients and solids from the water column, maximum periphyton productivity may be desirable. The highest periphyton productivity in this study occurred in HFW 5, which was the wetland with the shortest residence time. This may have contributed to periphyton productivity because nutrients remained available throughout the wetland, even close to the outflow. Epiphyte growth may also benefit from the presence of predominantly submerged species such as *Potamogeton* spp. or plants with a horizontal growth habit such as *Polygonum* spp. Unfavorable conditions include

low nutrient supply brought about by a low flow regime and both large and shallow depths. Depths greater than 70 cm resulted in areas of open water and therefore less surface area available to epiphyte growth. Shallow water depth (<20 cm) also resulted in less surface area available.

The presence of periphyton can serve as an indicator of available nutrients in the water column. Periphyton responded to nutrient concentration (higher chlorophyll at the inflows) and to nutrient loading (higher chlorophyll in the HFW). In addition, periphyton biomass on the samplers indicated areas of nutrient availability (HFW 3 and 5 outflows) and nutrient limitation (LFW 4 outflow).

5 Effects of Hydrology on Water Column Primary Productivity in Constructed Freshwater Wetlands

Introduction

A study of the daily pattern of dissolved oxygen measurements in the four experimental wetlands at the Des Plaines River Wetlands Demonstration Project was conducted to investigate and quantify the water column primary productivity of constructed freshwater wetlands. Productivity was measured as the daily flux of dissolved oxygen produced by the water column plants (phytoplankton, periphyton, and submerged macrophytes). Knowledge of oxygen fluxes provide insight into the cycling of other elements found in natural systems and is basic to our understanding of ecosystem function (Meyer and Edwards 1990) and redox conditions in wetland water and soils. Four wetlands at the site were subjected to two different flow rates (HFW and LFW), and the influence of water flow on productivity was investigated. Along with higher flow rates came a higher loading of nutrients (Chapter 2), and the water column community was expected to respond to water flow because of the community's rapid turnover rate. Data were collected during the 1990 - 1992 growing seasons in the four constructed marshes. Primary productivity estimates enabled estimates of the contribution of water column autotrophs to the ecosystem's primary productivity (water column + macrophytes) and investigation of the role of hydrology on water column productivity.

Methods

Dawn-dusk-dawn open water measurements of daily changes in dissolved oxygen were used to estimate water column productivity in the wetlands (Odum 1956; Penfound 1956; Odum and Hoskin 1958; Jervis 1969; Nixon and Oviatt 1973; Kemp and Boynton 1980; Mitsch and Kaltenborn 1980; Fontaine and Ewel 1981; Reeder and Mitsch 1989; Meyer and Edwards 1990). All of the submerged components of an aquatic system (phytoplankton, periphyton, and submerged

macrophytes) are included in dissolved oxygen measurements. Therefore, water column productivity refers to the oxygen productivity of all of these elements of the aquatic ecosystem.

Sampling

Daily changes in dissolved oxygen were measured at three sampling sites per wetland (3 of 16 permanent reference quadrats; HFW 3: 1, 10, 16; LFW 4: 3, 8, 16; HFW 5: 2, 7, 11; LFW 6: 5, 9, 16) on eight sampling dates from May through October 1990 and on five sampling dates from May through September 1991 using a YSI Model 51B dissolved oxygen meter. Sampling sites were near the inflow, middle, and outflow of each wetland (Figures 3 - 6). Water depth at these sites varied slightly during the growing seasons; but throughout the wetlands, water depth ranged from 10 to 85 cm. In 1992, samples were taken at the same three quadrats in Wetlands 3, 4, and 5 and two additional quadrats were also sampled in each wetland (HFW 3: 3, 15; LFW 4: 4, 12). Sampling was not done in LFW 6 because it was dry for most of the 1992 growing season.

Productivity calculation

Dissolved oxygen was converted to energy by multiplying by 15.06 kJ/g O_2, a conversion factor based on the energy required to yield a gram of oxygen in photosynthesis (Kormondy 1984). Solar radiation was measured by an Eppley Precision Pyranometer Model PSP with a 180^0 horizontal surface receiver in watts per square meter at the Des Plaines River site. Percent energy efficiency was calculated using the ratio of kilojoules produced to kilojoules of solar radiation on each sampling day.

Chlorophyll *a*

Oxygen readings were supplemented with measurements of chlorophyll *a* concentrations on eight dates in 1990 and 1991. Water samples (100 mL) were collected from approximately 10 cm below the water's surface at the same sites used for the dissolved oxygen measurements. Samples were shaken, divided in two, and filtered in 50-mL increments immediately after collection. The filters were frozen until the time of analysis and then placed in centrifuge tubes with 10 mL of 90 percent acetone and stored at 4 °C for 24 hr. Absorbance wavelengths at 750, 665, 630, and 645 were measured using a 10-cm path length in a Bausch and Lomb Model 600 spectrophotometer. Concentrations were calculated using Lorenzen's trichromatic equations (Parsons, Maita, and Lalli 1984).

Data analysis

The means for productivity estimates and chlorophyll *a* concentration for each wetland were compared in a one-way ANOVA, with each wetland as a treatment, for each sampling date and for all of the dates combined.

Results and Discussion

Dissolved oxygen changes

The maxima and minima for dissolved oxygen and temperature do not indicate clear differences among wetlands or between sampling years (Table 10). Dissolved oxygen values as high as 16.5 mg $O_2 \cdot L^{-1}$ were observed, and minima below 1 mg $O_2 \cdot L^{-1}$ were frequently measured in all wetlands. The largest dawn-to-dusk change in dissolved oxygen was in LFW 6 in 1990 when an algal bloom resulted in a change of 14.7 mg $O_2 \cdot L^{-1}$ between dawn and dusk.

The seasonal patterns of solar radiation, dissolved oxygen production, and average percent solar efficiency are shown in Figure 39. Each year, the average diel shift in oxygen concentration was of approximately the same magnitude (1990 average change = 6 mg $O_2 \cdot L^{-1}$; 1991 average change = 5 mg $O_2 \cdot L^{-1}$; 1992 average change = 7 mg $O_2 \cdot L^{-1}$). The data from 1990 showed two peaks in the diel oxygen change, in late June and early August, a seasonal pattern typical of algal productivity because of changes in nutrient availability and subsequent structural changes in the community (Van Raalte, Veliela, and Teal 1976; Sand-Jensen and Søndergaar 1981; Roos 1983; Wetzel 1983a; and Perrin, Bothwell, and Slaney 1987). In 1991, only the late June peak in dissolved oxygen change was detected. Peaks in average percent solar efficiency occurred in mid-August in both 1990 (0.6 percent) and 1991 (0.4 percent) (Figure 39c); however, the peak solar efficiency in 1992 was measured in September (0.7 percent).

Productivity estimates

Using a one-way ANOVA with each wetland as a treatment, significant differences among the wetlands' mean primary productivity were evident on two of the eight sampling dates in 1990, three of five sampling dates in 1991, and on one date in 1992 (Table 11). The differences were not consistent across the hydrologic treatments used in the first 2 years, although in 1991 and 1992 a LFW had the lowest average gross primary productivity and a HFW had the highest gross primary productivity.

The water column primary productivity results for these constructed wetlands lie within the same range as natural aquatic systems. Measurements at the Des Plaines River wetlands are comparable with or higher than those from lake studies (Mitsch and Kaltenborn 1980; Fontaine and Ewel 1981) and from a freshwater wetland in Ohio (Reeder 1990). Findings from the Des Plaines River wetlands confirm that planktonic communities become established quickly (Sand-Jensen and Borum 1991).

Water column chlorophyll *a*

Average chlorophyll *a* concentrations were only statistically different among the four wetlands on three sampling dates (Table 12). The differences are not

consistent with hydrologic conditions. When the data for all the sampling dates were combined to consider the influence of wetland position (inflow, middle, and outflow) on chlorophyll concentration, no significant differences among the sampling sites were found. While the differences are not statistically significant, the data for the interaction of hydrologic regime (high or low flow) and wetland position do reveal a consistent pattern. Chlorophyll *a* concentrations were greatest at the inflow, decreased in the middle, and decreased still more at the outflow (Figure 40). The differences among sites were greater in HFW than in LFW. Higher chlorophyll *a* concentrations at inflow sites may have been due to the higher nutrient levels near the inflow.

A significant positive linear relationship between chlorophyll *a* concentration and oxygen production estimates appeared, although the variability was extremely high ($r^2 = 0.06$, $n = 93$, $p = 0.02$). The weak r-square value corroborates other studies in which no relationship or only a weak relationship between chlorophyll *a* and water column metabolism was found (Fontaine and Ewel 1981; Reeder 1990).

Chlorophyll *a* was quite low in all of the wetlands, when compared with typical values from other studies. At a freshwater wetland adjacent to Lake Erie, the average peak chlorophyll *a* concentration was 120 $mg \cdot m^{-3}$ (Reeder 1990), while at the Des Plaines site, the average peak for all four wetlands was only 8 $mg \cdot m^{-3}$. However, productivity estimates at the two sites were quite similar. A productive planktonic community is thought to have an average chlorophyll *a* concentration of 15 $mg \cdot m^{-3}$ (Wetzel 1983a).

Because water column productivity at the Des Plaines site fits the expected levels for a productive aquatic system, but chlorophyll concentrations do not, a great amount of the dissolved oxygen in the water column probably comes from the photosynthesis of benthic or attached algae or from submerged macrophytes. At peak growth, periphyton in these wetlands may constitute as much as 65 percent of the total water column phototrophic community (Cronk 1992). The relative importance of periphyton coupled with the low water column chlorophyll *a* concentrations suggest that a large portion of the oxygen measured by this method may have been produced by periphyton and benthic algae rather than by suspended phytoplankton.

Hydrology and primary productivity

No clear relationship between water inflow and average annual primary productivity was observed in this study (Figure 41). In 1990, the highest productivity was in a HFW (5) and the lowest was in a LFW (4); however, the other two wetlands did not follow this pattern. During the second growing season, a relationship between water inflow rate and primary productivity did appear: The HFW had higher water column primary productivity than the LFW ($p < 0.10$). However, by 1992, this pattern had changed and a LFW (4) had the highest water column primary productivity. The variability in the results indicates that factors other than hydrology had more influence on water column productivity in these wetlands.

Role of water column autotrophs in wetland organic carbon production

Algal production was often as important as macrophyte net primary production (NPP) in these wetlands, and its measurement was essential to an overall picture of ecosystem productivity. Water column primary producers (attached and floating algae and submerged macrophytes) contributed from 10 to 67 percent of the estimated total net primary productivity of each wetland (Figure 42), while the remaining percentage was attributed to macrophyte productivity. The relative importance of water column primary productivity within these new wetlands indicates that algal communities respond quickly to wetland establishment. Therefore, a measurement of wetland algal communities is essential to an understanding of both productivity and nutrient dynamics in wetlands.

Three years of measurements

Our measurements of water column primary productivity showed no decline over 3 years, even as macrophytes became better established and the potential for light limitation in the water column increased. In fact, in 1992, the highest water column productivity was in the wetland (4) that also had the highest macrophyte productivity. The range of productivity values found at the Des Plaines River Wetlands is typical of a productive natural system. Therefore, these newly formed wetlands can support heterotrophic aquatic life and maintain aerobic and anaerobic processes necessary for nutrient transformations.

Table 10
Maxima and Minima of Dissolved Oxygen (DO), Change in DO, and Water Temperature at Des Plaines Wetlands from May 5 - October 7, 1990, May 16 - September 12, 1991, and August 5 - October 17, 1992

	High flow Wetland 3			Low flow Wetland 4			High flow Wetland 5			Low flow Wetland 6	
	1990	1991	1992	1990	1991	1992	1990	1991	1992	1990	1991
Max. DO, mg $O_2 \cdot L^{-1}$	13.9	15.0	12.9	12.2	9.3	14.5	15.0	12.8	15.1	16.5	13.2
Min. DO, mg $O_2 \cdot L^{-1}$	0.5	0.3	0.6	0.8	0.4	0.4	0.6	0.4	0.9	0.6	0.6
Max. Change, mg $O_2 \cdot L^{-1}$	8.0	12.1	10.6	6.6	7.3	12.6	11.2	9.4	14.1	14.7	10.6
Min. Change, mg $O_2 \cdot L^{-1}$	0.3	1.2	1.7	3.8	1.4	2.5	3.8	0.0	2.4	1.4	1.2
Max.Water Temp., °C	29	27	24	29	27	22	29	27	25	29	29
Min. Water Temp., °C	13	18	5	12	19	7	11	18	7	13	16

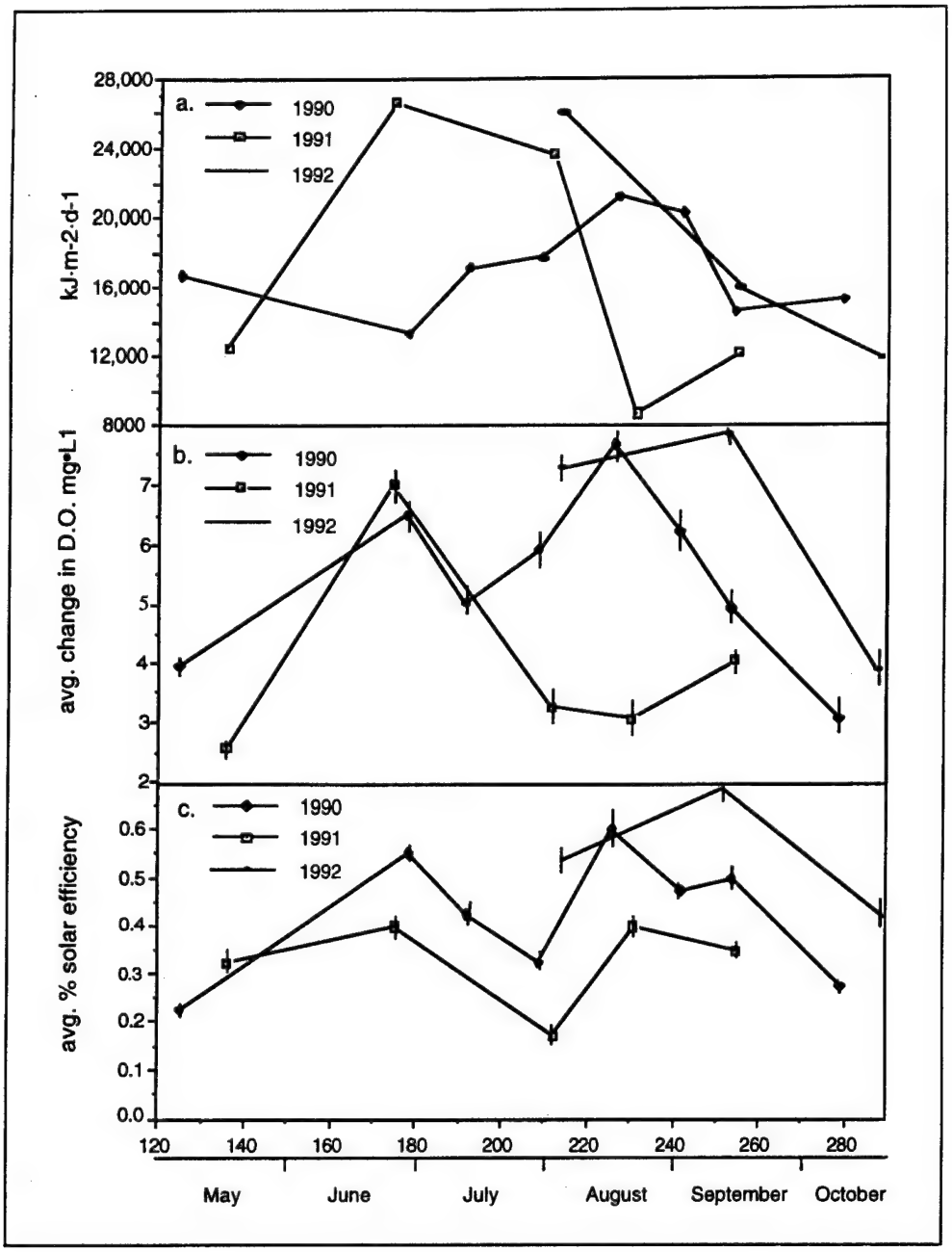

Figure 39. Des Plaines River wetlands measurements as a function of Julian date showing (a) solar radiation ($kJ \cdot m^{-2} \cdot day^{-1}$), (b) the average change in dissolved oxygen concentrations (mg $O_2 \cdot L^{-1}$) between dawn and dusk readings for each sampling date, and (c) average percent solar efficiency. Means are for all measurements on each date; bars indicate standard errors

Table 11
Gross Primary Productivity (average ± standard error) in All Wetlands

Date	High Wetland 3	Low Wetland 4	High Wetland 5	Low Wetland 6
	$kJ \cdot m^{-2} \cdot day^{-1}$			
5-May-90	49 ± 10 (a)	41 ± 14 (a)	27 ± 10 (a)	42 ± 16 (a)
27-Jun-90	75 ± 18 (a)	67 ± 15 (a)	81 ± 15 (a)	83 ± 43 (a)
11-Jul-90	46 ± 18 (a)	60 ± 15 (a)	121 ± 49 (a)	74 ± 28 (a)
28-Jul-90	53 ± 18 (a)	57 ± 11 (a)	73 ± 41 (a)	59 ± 36 (a)
14-Aug-90	87 ± 21 (b)	81 ± 11 (b)	193 ± 32 (a)	159 ± 45 (ab)
30-Aug-90	118 ± 28 (ab)	70 ± 11 (b)	119 ± 20 (ab)	139 ± 26 (a)
11-Sep-90	63 ± 32 (a)	51 ± 4 (a)	109 ± 14 (a)	100 ± 32 (a)
6-Oct-90	50 ± 22 (a) n = 3	38 ± 11 (a)	49 ± 14 (a)	47 ± 11 (a)
1990	68 ± 8 (bc) n = 23	58 ± 5 (c) n = 24	96 ± 13 (a) n = 24	88 ± 13 (ab) n = 24
16-May-91	28 ± 15 (a)	38 ± 8 (a)	46 ± 12 (a)	67 ± 41 (a)
24-Jun-91	183 ± 77 (a)	82 ± 8 (a)	119 ± 17 (a)	69 ± 38 (a)
31-Jul-91	57 ± 23 (a)	47 ± 11 (ab)	56 ± 4 (a)	15 ± 11 (b)
19-Aug-91	35 ± 10 (ab)	41 ± 9 (a)	57 ± 15 (a)	13 ± 1 (b)
12-Sep-91	48 ± 16 (ab)	69 ± 24 (a)	51 ± 26 (ab)	7 ± 3 (b)
1991	70 ± 21 (a)	56 ± 7 (ab) n = 15	66 ± 10 (ab) n = 15	34 ± 14 (b) n = 15
5-Aug-92	159 ± 34 (a)	160 ± 16 (a)	106 ± 28 (a)	
10-Sep-92	97 ± 17 (a)	150 ± 39 (a)	104 ± 4 (a) n = 4	
17-Oct-92	40 ± 10 (b)	78 ± 18 (a)	39 ± 6 (b)	
1992	98 ± 18 (ab) n = 15	129 ± 17 (a) n = 15	81 ± 13 (b) n = 14	

Note: Significant differences between wetlands revealed by ANOVA are indicated by unlike letters in parentheses ($p < 0.10$) (n = 3 in 1990 and 1991 and n = 5 in 1992 except where indicated).

Table 12
Chlorophyll *a* Concentrations in $mg \cdot m^{-3}$ (average ± standard error) in All Wetlands

Date	High Wetland 3	Low Wetland 4	High Wetland 5	Low Wetland 6
Jul-11-90	1.8 ± 0.6 (a)	2.3 ± 0.7 (a)	1.3 ± 0.3 (a)	4.4 ± 2.2 (a)
Aug-15-90	1.6 ± 0.9 (a)	2.8 ± 2.2 (a)	4.6 ± 2.3 (a)	3.1 ± 1.4 (a)
Aug-30-90	3.8 ± 0.5 (a)	1.0 ± 0.4 (b)	3.1 ± 1.2 (a)	3.1 ± 1.4 (ab)
Oct-06-90	2.5 ± 0.8 (a)	0.5 ± 0.3 (b)	0.6 ± 0.2 (b)	0.4 ± 0.2 (b)
May-16-91	4.8 ± 0.9 (a)	2.4 (*) n = 1	3.1 ± 0.7 (a)	4.6 ± 0.6 (a)
Jun-24-91	15.1 ± 2.7 (a)	2.7 ± 0.2 (a)	11.4 ± 9.1 (a)	3.9 ± 0.5 (a)
Jul-31-91	4.8 ± 1.0 (ab)	3.1 ± 0.5 (b)	9.2 ± 3.4 (ab)	9.3 ± 3.0 (a)
Aug-19-91	6.7 ± 3.0 (a)		3.4 ± 0.6 (a)	9.2 ± 3.6(a)

Note: Significant differences on each date that were revealed by ANOVA are indicated by unlike letters ($p < 0.10$) ($n = 3$ except where indicated).
* not included in ANOVA because only one data point was available.

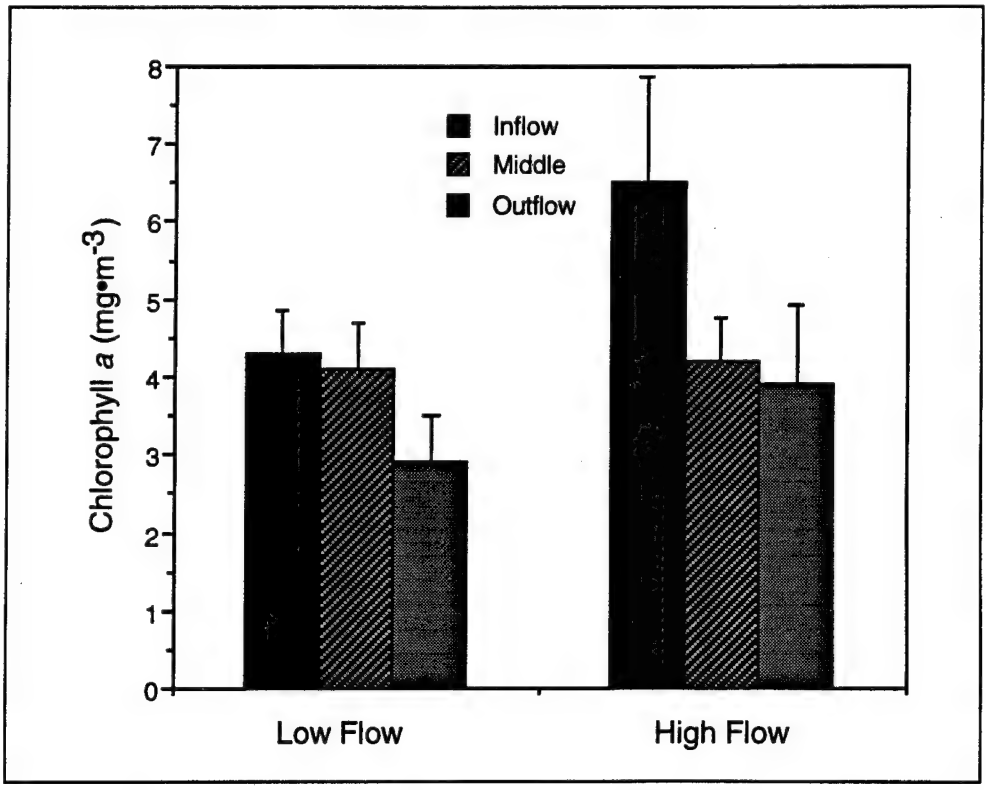

Figure 40. Average chlorophyll *a* concentrations ($mg \cdot m^{-3}$) ± standard error at inflow, middle, and outflow stations in high-flow wetlands (3 and 5) and low-flow wetlands (4 and 6) (Data are from 1990 and 1991. None of differences were significant at $p < 0.10$)

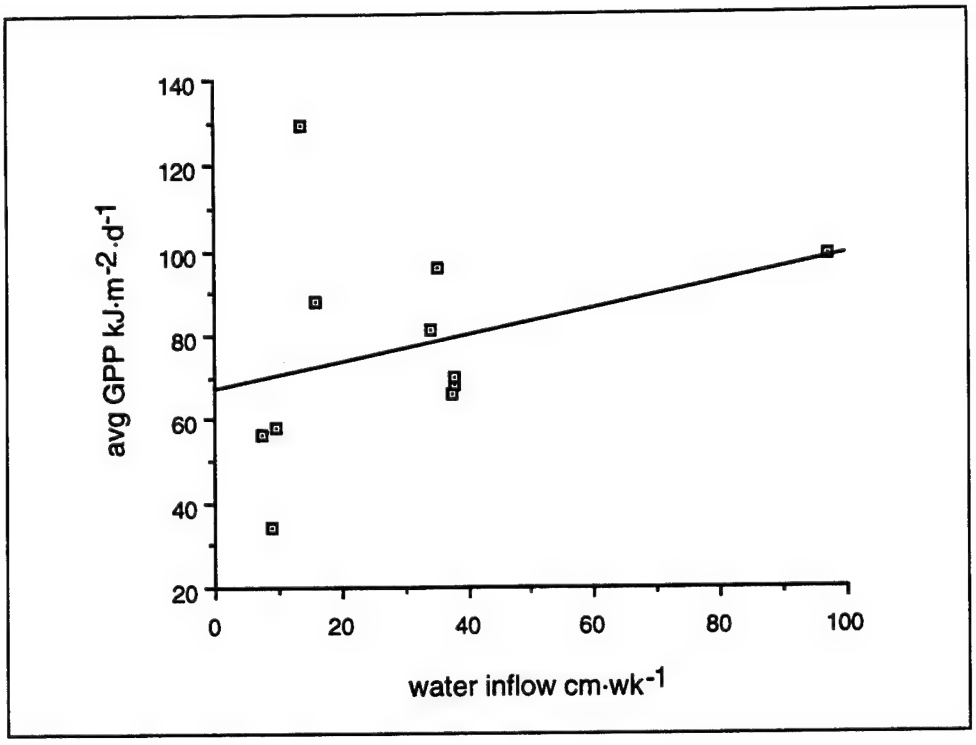

Figure 41. Average gross primary productivity ($kJ \cdot m^{-2} \cdot day^{-1}$) as a function of water inflow ($cm \cdot week^{-1}$)

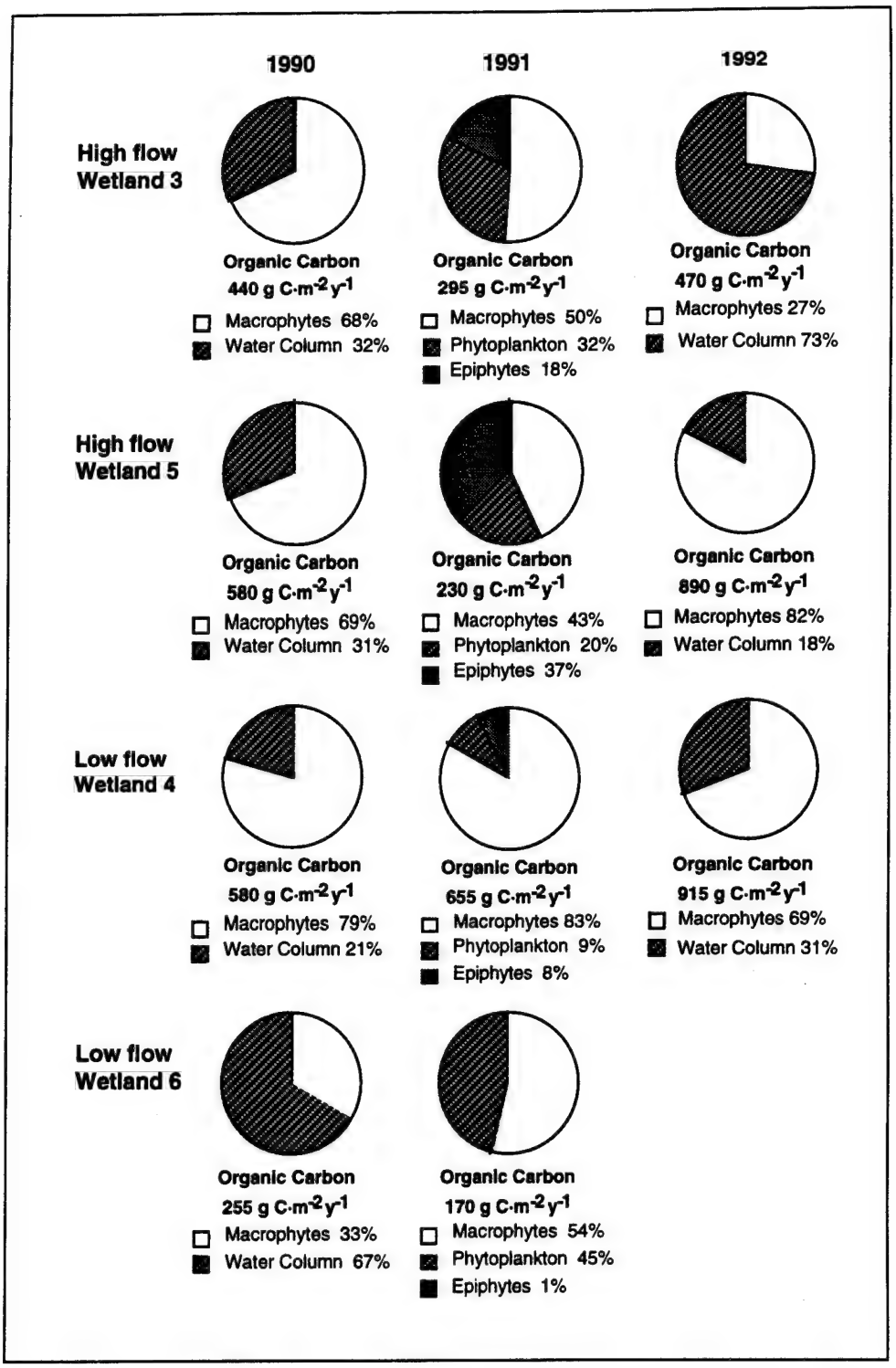

Figure 42. Estimated carbon production (g $C \cdot m^{-2}$of wetland·$year^{-1}$) by aboveground portion of macrophytes (Chapter 3) and water column primary producers (as measured by the dissolved oxygen method) and percent contribution of each component to net carbon of the wetlands from 1990 -1992 (In 1990 and 1992 water column production percentages included epiphytes, while in 1991, contribution of epiphytes to water column productivity was estimated in a concurrent study (Chapter 4))

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Wlliam J. Mitsch, Julie K. Cronk

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13. ABSTRACT *(Maximum 200 words)*

Four 2- to 3-ha constructed freshwater riparian wetlands in Lake County, Illinois, were subjected to two hydrologic regimes of pumped river water to simulate nonpoint source pollution. The experimental wetlands at the Des Plaines River Wetland Demonstration Project were designed to develop and test wetland design principles, construction methods, and management programs needed to create and maintain wetlands for the purposes of water quality management, flood control, and fish and wildlife habitat. High-flow wetlands (HFW) with short retention times received 34 to 38 cm of river water per week, and low-flow wetlands (LFW) with high retention times received 10 to 15 cm per week. This report summarizes research results for phosphorus dynamics and retention, macrophyte development, periphyton productivity, and overall water column metabolism through 1992. All of these functions were hypothesized to be related to hydrologic conditions.

Measurements of soluble reactive phosphorus and total phosphorus were made weekly at the inflow and outflow of each constructed wetland from 1990-1992 and at permanent reference quadrats throughout the wetlands in 1991. Intensive sampling with six samples per day was also carried out in 1992 on two of the wetlands. The wetlands retained over 60 percent of incoming phosphorus with general improvement from the first to second year.

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In the third year, inflow to one of the wetlands more than doubled, and phosphorus retention decreased from 85 to 53 percent. Spatial patterns of phosphorus concentrations indicate that in the LFW, over 60 percent of the incoming phosphorus was removed from the water column within 15 to 20 m of the inflow. In the HFW, phosphorus decreased gradually from inflow to outflow with 60-percent removal within 70 to 100 m of the inflow. Intensive sampling in 1992 with over 800 analyses of phosphorus from the outflows of two wetlands showed an 81-percent decrease in phosphorus concentration from inflow to outflow in the LFW and a 74-percent decrease in the HFW. Intensive sampling also showed an 87-percent decrease in phosphorus mass from inflow to outflow in a LFW and 83-percent decrease in phosphorus mass in a HFW. Analysis of other processes in the wetlands illustrated that most of the phosphorus was probably retained through sedimentation of inorganic sediments, with macrophyte uptake accounting for a substantial sink of phosphorus. Periphyton and planktonic uptake was less significant.

Macrophyte biomass production and successional changes in wetland species composition were monitored over 5 years, from 1988 through 1992. From dry conditions in 1988 to flooded conditions beginning in 1989, the species composition of each wetland's plant community changed to include almost 100-percent wetland species soon after flooding. The number of species the four wetlands had in common decreased from 1988 to 1989 as the plant community adjusted to flooded conditions and then increased as the wetlands developed and the plant communities converged. Biomass increased from 1989 to 1990 in all of the wetlands as the plant communities changed from upland to wetland species. After 1990, biomass increased progressively each year in LFW. HFW showed more variable patterns from year to year. Peak biomass of emergent macrophytes ranged from 250 g dry weight/m^2 in one HFW to 1,470 g dry weight/m^2 in another HFW after 4 years of flooding. Emergent macrophyte productivity was determined to be more a function of antecedent conditions and water levels than flow-through conditions over this relatively short time.

Periphyton samplers were placed near the inflow and outflow of all four wetlands, and dry weight, organic dry weight, and chlorophyll *a* was measured every 2 weeks from May through August 1991. Periphyton growth on macrophytes was estimated three times in all four wetlands. Periphyton chlorophyll *a* on artificial surfaces was higher in the HFW than in the LFW. Samplers in one HFW (HFW 5) had higher dry weight and organic dry weight than the other wetlands. Inflow samplers had higher chlorophyll *a*, dry weight, and organic dry weight than outflow samplers ($p < 0.10$). These data suggest that periphyton responded to the hydrologic treatment as well as to position within the wetland. Of the three macrophyte species examined for epiphytes (*Polygonum* spp., *Phalaris arundinacea*, and *Typha* spp.), epiphyte dry weight, organic dry weight, and chlorophyll *a* were highest on *Polygonum* spp. and lowest on *Typha* spp. Net epiphyte biomass was estimated to be highest in a HFW (HFW 5).

Diurnal changes in dissolved oxygen concentrations were used during three growing seasons to estimate water column primary productivity. Productivity was generally higher in the HFW (84 $kJ \cdot m^{-2} \cdot day^{-1}$ in 1990; 69 $kJ \cdot m^{-2} \cdot day^{-1}$ in 1991) than in the LFW (73 $kJ \cdot m^{-2} \cdot day^{-1}$ in 1990; 46 $kJ \cdot m^{-2} \cdot day^{-1}$ in 1991). In 1992, LFW 4 had the highest water column primary productivity (129 $kJ \cdot m^{-2} \cdot day^{-1}$). Chlorophyll *a* concentration did not correlate to productivity. Water column primary producers contributed an estimated 17 to 67 percent of the net aboveground carbon production of each wetland, but no consistent pattern of increased or decreased importance of water column productivity over 3 years was observed.